SALDANHA BAY WQT Water Quality Trust

THE STATE OF SALDANHA BAY AND LANGEBAAN LAGOON 2026

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THE STATE OF
SALDANHA BAY AND LANGEBAAN LAGOON
2026

Report prepared for:

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FOREWORD

The Saldanha Bay Water Quality Trust has been active for 27 years, representing almost three decades of dedicated environmental monitoring, research, and scientific investigation in the Saldanha Bay region. Over this period, the Trust has developed an extensive body of knowledge and a valuable long-term environmental dataset that forms the foundation of this report.

The Trust takes great pride in its academic character and is committed to conducting its work according to rigorous scientific principles. Our approach is founded on sound methodology, consistent long-term data collection, scientific integrity, and the objective analysis and communication of environmental information.

This report is the product of many years of research and the systematic collection and analysis of scientific data. It provides a scientifically informed assessment of the health and condition of Saldanha Bay and the Langebaan Lagoon, while contributing to the broader discussion on balancing regional development with environmental stewardship.

The region faces increasing challenges, including climate change, habitat loss, pollution, and declining biodiversity. The Trust's long-term research provides an important scientific foundation for understanding these pressures and supporting informed decision-making, effective conservation, and sustainable development.

Jacques Bezuidenhout
Chairperson
Saldanha Bay Water Quality Trust

Figure 1. The 2026 Trustees. From left to right: Adele Groenewald (TPT), Pierre Nel (SANP), Nazeema Duarte (SBM), Jacques Bezuidenhout (Chairman), FC van Wyk (Metsep), Elmien de Bruyn (Duferco), Ethel Coetzee (TNPA).

ACKNOWLEDGMENTS

Many people and organisations have contributed to the development and production of the State of the Bay report over the years. Most important are those that generously fund the work of the Water Quality Trust (WQT) (the Trustees, see Table 1 below). Those that have made their data available for use in the production of the report (see Table 3 below) and also those that have provided logistical and other support over the years in facilitating field surveys and data collection (Table 2). A special thanks to the various contributing photographers, who have given permission to use their work in general (including Steven Benjamin and Aiden Biccard); individual photography credits are otherwise provided in the relevant figure captions.

A report of this magnitude is only made possible through the contributions of a large and diverse team, which includes field scientists, Research Technicians and Research Assistants. To this end, we would particularly like to acknowledge the efforts of the benthic taxonomy team, Mpho Malgas, Olwethu Nkwentsha, Emihle Joja and Dr Jyothi Kara for their contribution to this work.

Table 1. Trustees of the WQT (2026).

Organisation	Organisation
Transnet National Ports Authority	Duferco Steel Processing
SAPO Transnet Port Terminals	Premier Fishing
Black Mountain (Vedanta)	Sunrise Energy
Sea Harvest	Club Mykonos
Saldanha Bay Municipality	ArcelorMittal Steel
Tronox	PPC
SFF	Langebaan Country Estate
Kropz-Elandsfontein	Improchem
Stellenbosch University	Sishen Iron Ore Company (Pty) Ltd
OiltankingMOGS Saldanha	Anglo American
Postberg (Pty) Ltd	
Friends of the Trust	
Indawo	Poop Haul (Pty) Ltd
BSASA	Portfolium Fund Services
The Cove Home Owners Association	

Table 2. Organisations and individuals who have provided logistical and other support over the years in facilitating field surveys and data collection for the State of the Bay reports.

Organisation	Organisation
Langebaan Yacht Club	Special Forces Regiment
SAS Saldanha	SANParks West Coast National Park

Table 3. Organisations and individuals who have generously made data available for compilation of the State of the Bay reports.

Organisation	Contribution
Animal Demography Unit of the University of Cape Town	Summer and Winter counts of shorebirds in Langebaan Lagoon
Bivalve Shellfish Farmers Association of South Africa (BSASA)	Production data for mariculture operators in Saldanha Bay
Council of Scientific & Industrial Research (CSIR)	Data on sediment quality and water quality in Saldanha Bay and Langebaan Lagoon, data from the wave rider buoy
Department of Forestry Fisheries and the Environment (DFFE)	Data on trace metals in shellfish from Saldanha Bay (farmed and wild caught shellfish), data on seabirds and seals on the offshore islands, water quality data from the Aquaculture Development Zone (ADZ) monitoring programme, production data for mariculture operators in Saldanha Bay, pelagic fish stock distribution, seal pup count data
Department of Water & Sanitation (DWS)	Data on ground water level and quality from aquifers in the Saldanha Bay area
Krops-Elandsfontein	Data on ground water level and quality from aquifers in the Saldanha Bay area
Saldanha Bay Municipality	Data on the Saldanha and Langebaan Waste Water Treatment Works, groundwater monitoring data, beach profile data.
Sea Harvest	Effluent and receiving environment quality data from their fish processing plant
Prof. Tammy Robinson (Stellenbosch University)	Alien species detection and research
West Coast District Municipality	Data on bacterial indicators in coastal waters, data on groundwater
Pierre Nel (SANParks West Coast National Park)	Data on visitor numbers, BRUV (Baited remote underwater video) data
Transnet National Ports Authority	Data on ore exports, shipping traffic and ballast water discharges, effluent and receiving environment water quality data from the reserve osmosis desalination plant
Assoc. Prof. Colin Attwood (University of Cape Town)	Beach seine net data from surveys conducted at Geelbek, Langebaan Lagoon.

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EXECUTIVE SUMMARY

Introduction

Regular, long-term environmental monitoring is essential to identify and enable adaptive management and mitigation of negative human impacts on the environment, 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 in South Africa, while also supporting diverse and important marine and coastal biodiversity, together with a range of ecologically and socioeconomically important activities, including tourism and fishing. While the development of the Saldanha Bay port has significantly altered many of its environmental components, all developments within the area (industrial, residential, tourism, etc.) have the potential to negatively impact ecosystem health. Continued environmental assessment is therefore critical for detecting changes in environmental condition, distinguishing natural variability from anthropogenic impacts, and informing appropriate management responses to ensure the continued ecological and socioeconomic value of the Saldanha Bay/Langebaan Lagoon system.

Monitoring was initiated by what was previously known as the Saldanha Bay Water Quality Forum Trust (SBWQFT) in 1996. This voluntary organization, renamed as the Saldanha Bay Water Quality Trust (WQT), represents various organs of State, local industry and other relevant stakeholders and interest groups. Since its inauguration, the WQT 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. This ecosystem monitoring culminated in the "State of the Bay" report series, which began in 2006, and has been produced annually since 2008. This report series serves 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 system. Monitoring of water quality, sediment quality and benthic macrofauna has been undertaken since 1999, with monitoring of surf zone fish and rocky intertidal macrofauna added to the programme in 2005. Several additional indicators have been incorporated into the monitoring programme over the years, including a summary of activities and discharges that may affect system health, management and policy developments, groundwater resources in the surrounding areas, the presence and spread of alien and invasive species, coastal erosion, and changes in the extent of aquatic plants (macrophytes) that perform important ecological functions within estuarine habitats. Given the results of more than 15 years of historical annual monitoring, and in accordance with the new approved 10-year monitoring plan for the WQT, many parameters are now monitored annually or biannually.

This report is the 19th of the series. It presents the 2026 data in the context of work commissioned by the WQT, historical trends pre-dating the State of the Bay series and a range of other scientific monitoring programmes and studies. Where possible, trends and areas of concern have been identified and recommendations for future monitoring are presented, with a view to further improving environmental management and monitoring in the area. Despite noteworthy successes in environmental monitoring, and the conservation of important marine habitats within the West Coast National Park Marine Protected Area, historical and on-going development of the area has resulted in environmental degradation of some marine and coastal habitats. Management interventions have contributed to significant improvements in ecosystem health; however, continued monitoring and management remain essential to support the rehabilitation

of degraded habitats, identify and mitigate potential impacts associated with future development, and sustain the ecological integrity and associated benefits of the system.

Management and policy development

The current and future desired state of the greater Saldanha Bay area is polarised, with industrial development (such as the Saldanha Bay Industrial Development Zone and associated industries), residential areas, and conservation areas (Ramsar Site, Marine Protected Areas, and National Parks) located directly alongside one another. The area also supports multiple, often conflicting industries, including port operations, fisheries, mariculture, tourism and recreation, and the natural environment itself.

This highlights the need for sustainable development. This partly requires effective spatial planning and management. Development, activities, and land use in the area are guided by a range of national, provincial, and local legislation, as well as municipal-specific spatial plans, frameworks, and management instruments. This Chapter provides an overview of the legislative framework relevant to the system, including key updates and amendments made during 2026 to national, provincial and local legislation and subordinate legislation. If effectively implemented, these instruments offer a strong foundation for balancing conservation needs with the pressures of rapid development in the Bay area. A key summary of relevant legislation is provided below:

- National: The principal national environmental legislation includes the National Environmental Management Act, 1998 (Act 107 of 1998); National Environmental Management: Protected Areas Act, 2003 (Act 57 of 2003); National Environmental Management: Air Quality Act, 2004 (Act 39 of 2004); and National Environmental Management: Integrated Coastal Management Act, 2008 (Act 24 of 2008). Collectively, these provide the legislative framework for environmental assessment and authorisation, protected area management, air quality management, and the management of coastal and marine resources.
- Provincial: The Western Cape Biodiversity Act, 2021 (Act 6 of 2021), 2023 Western Cape Biodiversity Spatial Plan, and Western Cape Land Use Planning Act, 2014 (Act 3 of 2014) provide the framework for biodiversity conservation and the integration of environmental considerations into spatial planning and development.
- Local: The Saldanha Bay Municipality Integrated Development Plan (2025/26), covering the 2022–2027 planning period, and the Saldanha Bay Municipal Spatial Development Framework (2025–2030) provide the current framework for municipal development and spatial planning. The Greater Saldanha Bay Strategic Environmental Assessment (2019) further identifies key environmental risks and provides mitigation measures and guidance for sustainable development in the area.
- International: South Africa is party to a range of international environmental agreements, including the Ramsar Convention on Wetlands of International Importance (1971), Convention on Migratory Species (Bonn Convention), and Convention on Biological Diversity (1992). These commitments, including the Kunming-Montreal Global Biodiversity Framework (2022), provide an overarching framework for biodiversity conservation, ecosystem restoration and sustainable development relevant to the management of the Saldanha Bay/Langebaan Lagoon system.

Over the last few years, new and revised regulations, guidelines, and policies have been published under various Acts relating to climate change, air quality, coastal management, waste, water, marine spatial planning, marine resources, and land use. Furthermore, new laws such as the Electricity Regulation Amendment Act (Act 38 of 2024) and the Western Cape Land Use

Planning Amendment Bill [B 3–2025] have been enacted, while several draft Bills and regulations are currently under consideration. National dust control regulations were gazetted in March 2026, repealing the previous 2013 National Dust Control Regulations, and an updated National Biodiversity Assessment (NBA) was released in late 2025. The State of Knowledge and Situation Assessment Report 2025 has been completed for the West Coast National Park, along with the stakeholder processes for the draft West Coast National Park Marine Protected Areas, Islands and Langebaan estuarine lagoon management plan (2027-2031). The coastal management line for West Coast National Park is also currently under revision.

Activities and discharges

Anthropogenic pollutants and wastes find their way into the Bay from a range of activities and developments. These include port operations, shipping, ballast water discharges and oil spills, export of metal ores, municipal (sewage) and industrial discharges, biological waste associated with mariculture and storm water runoff. Urban and industrial developments have encroached into coastal areas and development of the port is expected to increase dramatically with the establishment and commissioning of the Saldanha Bay Industrial Development Zone. This chapter looks at annual trends in many of the activities and operations around the Bay and Lagoon which may impact the surrounding marine environment.

The human population surrounding Saldanha Bay and Langebaan Lagoon was estimated at 125 687 individuals in 2022 and is projected to reach 170 100 by 2027. Numbers of visitors to the West Coast National Park (WCNP) peak in the summer months and particularly the December holiday period, but also during the flower season in August and September. Since the significant drops in 2020 and 2021 due to the COVID-19 lockdowns, overall numbers of international visitors to South Africa increased more than three-fold in 2022, increased again in 2023, and again in 2026. This rapid population growth and seasonal tourism influxes translate to corresponding increases in the amount of infrastructure required to house and accommodate these people, and in the amount of waste and wastewater that is produced which must be treated and disposed of.

Metal ores exported from the Port of Saldanha Bay include iron, lead, copper, zinc, and manganese. Most of the iron ore is exported from the Iron Ore Terminal. Overall metal export volumes increased from 51.2 million tonnes in 2024/2025 to 55.4 million tonnes in 2025/2026. Metal exports from the Multi-Purpose Terminal increased over the last five years, driven mainly by increased zinc exports, but have declined over the last two years to 327 758 tonnes in 2024/2025. Manganese exports from the Saldanha Port increased in 2024/2025 to 6.35 million tonnes. Lead exports have averaged approximately 56 436 tonnes a year over the last decade. Copper is exported in small quantities compared to other metal ores, with exports in 2024/2025 also dropping relative to the previous year (to 11 840 tonnes per year), below the decade average.

Dredging in Saldanha Bay has had tremendous immediate impact on the health of the Bay. Extensive dredging has been conducted in the Bay in the past, and particles suspended in the water column during dredging impact components of the ecosystem in a range of ways. However, the damage caused by dredging is generally reversible in the long term, and although the particle size composition of the settled material is likely to be different, ecological functioning as well as major species groups generally return in time. In June 2025, the Port completed dredging operations at the Mossgas quay, with a total volume of approximately 20 000 m^3 of material removed. No further dredging has taken place since then.

The total number of ships entering the Port of Saldanha nearly tripled between 1994 and its peak in 2018/19 (261 to 616 vessels per annum). This, along with increases in average vessel size,

more than tripled the volume of ballast water discharged annually between 1994/5 and 2020/2021 (from 8.2 to 25.9 million m^3). In 2025/2026 however, there was a decline in the discharge ballast volume (down to 17 million m^3), some 30% below the average of 23.2 million m^3 for the decade prior to 2020, despite the overall increase in vessel traffic.

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, and industrial effluent (brine, cooling water discharges and fish factory effluent). Before the 2017/18 provincial drought, sewage discharge was arguably the most significant waste product that was discharged into Saldanha Bay in terms of its continuous environmental impact. Sewage is harmful to biota due to its high organic content and high nutrients levels which can stimulate primary production that, in turn, can lead to changes in species composition, decreased biodiversity, and toxicity effects. The Saldanha Bay Municipality has made improvements both to existing wastewater treatment facilities and in the management of the effluent discharged from the plants. Nowadays, most of the wastewater is being used for irrigation.

Favourable environmental conditions of Saldanha Bay have facilitated the establishment of an aquaculture industry. The development of this sector was identified as a key priority of Operation Phakisa as a means of job creation and the local economy. As a result a sea-based Aquaculture Development Zone (ADZ) was established in 2018. The total area available for aquaculture in the Bay is a maximum of 884 ha within four precincts (Small Bay, Big Bay, Outer Bay North and Outer Bay South). As of September 2025, there are 30 total allocated rights (namely for bivalve farming), with 24 actually operational at present. 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 (see Water Quality chapter). The Department of Forestry, Fisheries and the Environment (DFFE) in conjunction with others are continuously monitoring the ADZ precincts, conducting ongoing research to ensure that any negative impacts of the aquaculture are identified early, and appropriate action is taken. A substantial body of work (some of which is discussed in this report) has been undertaken in compliance with the stipulations in the EA and the Environmental Management Plan for the Saldanha Bay ADZ. These include water quality monitoring, dispersion modelling, baseline macrofauna and physicochemical surveys (2020), benthic macrofauna monitoring surveys (2021 and 2024) and annual benthic chemical surveys (2019–2025), a survey of the hard substrate/reef structure in Big Bay (2022, 2025), dispersion model validation and a report on the first qualitative sampling of fouling organisms (2022).

Generally, these ecological monitoring results indicate few significant environmental effects of the farms, within the precincts, in a given year. The only exception being the areas under the mussel rafts in Small Bay. Total Organic Carbon and Total Organic Nitrogen, sediment chemistry, Particle Size distribution, and infaunal community indices indicate these areas have been negatively impacted from a long history of bivalve culture. Therefore, it is suggested that some of rafts should be allowed to 'lie fallow' without active farming on a rotational basis.

Groundwater

This chapter provides an update on the municipal wellfields, including groundwater level and quality monitoring (electrical conductivity) from the Department of Water and Sanitation (DWS) borehole monitoring network over the past year for the Greater Saldanha region. The chapter also provides a a review of the Elandsfontein mine's groundwater monitoring cross-section, which tracks the mine's dewatering and recharge system relative to the paleochannel pathway feeding Langebaan Lagoon. Saldanha Bay Municipality, in partnership with GEOSS South Africa (Pty) Ltd,

maintains a dedicated groundwater monitoring network at the municipal wellfields, complemented by an extended network of DWS boreholes. The network has remained unchanged over the past year, ensuring continued spatial coverage for effective aquifer management.

Abstraction at the Langebaan Road and Hopefield wellfields is monitored daily and reported monthly. Licensed volumes have not been exceeded, with only 8.84% and 2.2% of the respective annual allowances abstracted to date in 2026, demonstrating ongoing compliance and responsible abstraction. However, the recent low abstraction volumes can, in part, be attributed to extensive vandalism at both wellfields in September 2025 so this needs to be considered.

The Saldanha Bay groundwater system comprises an upper, generally unconfined aquifer and a lower aquifer. Hydrochemical data indicate that the upper aquifer has higher mineralisation, while the lower aquifer, hosting the wellfield boreholes, shows lower conductivities, except for borehole G46092, outside the main flow path. The bedrock aquifer exhibits the greatest variability, reflecting diverse water-rock interactions and recharge conditions.

Long-term monitoring indicates stable groundwater levels and consistent water quality across the network, suggesting current abstraction and other groundwater uses are not stressing the aquifer systems. Boreholes at Geelbek, near the southern tip of Langebaan Lagoon, show no evidence of impact from upgradient abstraction or mining: water levels here remain within centimetres of pre-mining conditions, and although electrical conductivity has shifted more than elsewhere in the network, the change is towards fresher rather than more saline water. A dedicated monitoring cross-section running from Geelbek across the Elandsfontein mine reinforces this, showing that pit dewatering has produced a well-contained cone of drawdown around the mine itself, recovering to under a metre within a few kilometres in either direction, with no evidence of this influence extending towards the lagoon. Together, these results support the continued protection of this sensitive ecosystem.

Water quality

Water quality is monitored to better understand changes in the environmental health of the Bay. Regional oceanographic processes are important drivers of natural variation in water temperature, salinity, dissolved oxygen and nutrient concentrations within Saldanha Bay. These natural processes interact with effluent discharges and other anthropogenic activities and developments in the Bay, contributing to changes in water quality, particularly within Small Bay, where water exchange is more restricted.

After the construction of the Iron Ore Terminal and the Marcus Island causeway, the retention time of water in Small Bay increased, with enhanced clockwise circulation and increased current strength alongside these structures. As a result, 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. Furthermore, the wave energy climate (significant wave heights) in the coastal region of Saldanha Bay has increased over time (1960 to present day), supporting well-documented observations of changing weather and climate around Southern Africa. This should be considered when assessing, and forecasting, changes in oceanographically driven aspects of water quality.

Hypoxic events (periods of low dissolved oxygen <2ml/l) in Small Bay were first detected after the major harbour development in the 1970s, but the frequency of such events does not appear to have changed much in recent times. Results affirm that natural hypoxic conditions in the lower

part of the water column are frequent during the summer-autumn season in both Big Bay and Small Bay which is caused by upwelling related processes and thermocline development. In Small Bay, anthropogenic organic loading appears to exacerbate the situation, with decreased dissolved oxygen measured at sites under mariculture farms, suggesting this decrease in oxygen availability is associated with both organic pollution and limited flushing.

Regular, long-term monitoring of microbiological indicators at 20 stations around the Bay, including recreational beaches, indicates a substantial improvement in faecal pollution since the early monitoring record. The 2026 *Escherichia coli* (*E. coli*) results were the best recorded to date, with 19 of 20 sites rated as having 'Excellent' water quality. Several beaches that declined in quality in 2025, including Langebaan Main Beach, Leentjiesklip and Tooth Rock, returned to 'Excellent' status in 2026. Bok River continues to record the highest *E. coli* counts, while Hoedjiesbaai maintained its 'Excellent' recreational water quality rating for a second consecutive year. The increased reuse of treated wastewater, historically discharged via the Bok River Mouth, appears to be contributing to improved water quality. Water quality in Small Bay remains more variable, particularly in relation to mariculture guidelines, with nine of 10 sites exceeding the compliance thresholds in 2026. Although sustained improvements in compliance have occurred at several sites, elevated faecal coliform levels remain a concern for mariculture, particularly at sites influenced by the Bok River Mouth. In Big Bay, three of four sites exceeded the mariculture thresholds, which is a decline relative to 2025. Given the importance and anticipated growth of the mariculture and tourism sectors, continued investment in wastewater and stormwater infrastructure and measures to reduce microbial pollution remain important.

Concentrations of trace metals in shore-based and farm-based filter feeding bivalves (mostly mussels and oysters) in Saldanha Bay have historically been monitored by the DFFE, annual State of the Bay surveys, and by mariculture farm operators. In 2026, concentrations of trace metals in mussel tissue samples were below prescribed limits for most metals, with only arsenic levels being exceeded. Lead concentrations remained below the guideline limit which suggests conditions in the Bay may be improving. Lead levels in mussels and oysters from mariculture farms remain lower than those from shore-based samples, and in nearly all cases are within the limits of the relevant food safety guidelines. Lead levels in mussels and oysters collected from the shoreline have regularly exceeded the guideline limits at various locations, suggesting localised terrestrial sources such as road runoff and/or longer bioaccumulation times in wild shellfish compared to farmed stock. Historically, arsenic concentrations regularly exceeded food safety guidelines from both shore-based and farmed bivalves, and this is again the case with the 2026 data. Note, however, that only one sample was collected from farmed oysters in 2025/2026, creating an incomplete picture for this species (although this sample did exceed arsenic concentration limits). This, in combination with elevated concentrations above recommended limits in mussels collected from the shore remains a concern.

Overall, the 2026 water quality results present a mixed picture. Recreational water quality remains good, with substantial improvement in microbiological indicators over the longer monitoring record. However, there has been a slight decline in compliance with mariculture water quality limits compared with previous years, with exceedances particularly evident in Small Bay. Continued monitoring, together with ongoing improvements in effluent, stormwater and metal-ore discharge management, will be important to maintain these improvements and protect the Bay's environmental health and its recreational and commercial uses.

Coastal erosion

The 2026 coastal erosion assessment continued the remote-sensing-based monitoring programme, with Sentinel-2 imagery and the CoastSat workflow algorithms used to assess recent shoreline-changes and seasonal patterns along Langebaan, Leentjiesklip, Calypso, Mykonos and Paradise beaches. The assessment also examined the relationship between seasonal shoreline movement and significant wave height, providing greater insight into the physical processes contributing to erosion and accretion.

Most of Leentjiesklip, Calypso and Mykonos beaches have remained relatively stable or experienced low rates of accretion of approximately 0–1 m/year. More pronounced changes occurred around the Langebaan groynes, where the shoreline south of the structures have been advancing by approximately 15.8 m/year, while the section between the groynes continues to erode at approximately 2.3 m/year. Seasonal analysis revealed that the erosion hotspot was most vulnerable during winter, when the shoreline retreated by an average of almost 12 m from its summer position. These findings indicate that, although the groynes have promoted sediment accumulation and reduced erosion in some adjacent areas, they have not stabilised the entire coastline in this area.

Increases in offshore wave height were linked with coastal erosion observed along Langebaan North Beach and much of central Paradise Beach, with south-westerly waves most frequently linked to the areas shoreline retreat. In contrast, Langebaan South Beach generally accumulated sediment during periods of higher energy winter waves, suggesting that sediment redistribution processes operating within the sheltered lagoon may be more influential in this area. Although most of Paradise Beach showed relatively low annual rates of change, continued erosion of the central beach, dunes and exposed calcrete remains an important concern because it may progressively weaken the coastline's natural protection against future storms.

Bi-annual monitoring is recommended, supported by in situ wave and atmospheric measurements, beach profiling and higher temporal resolution assessments before and after major storm events. It is also recommended that a CoastSnap community science programme be established at Langebaan North and South beaches to provide more frequent shoreline observations, address gaps in satellite coverage and encourage public participation in long-term coastal stewardship.

Benthic systems

Soft-bottom infaunal benthic macrofauna (animals living in the sediment larger than 1 mm) are frequently used as indicators of marine environmental health, particularly in response to human impacts. These species are typically short-lived and, consequently, their community composition often responds rapidly to environmental change. For example, monitoring has shown that dredging events in 2012, 2015, and 2017 led to a decline in macrofaunal abundance and biomass, followed by rapid recovery. In each instance, these disturbance events resulted in a decrease in filter feeders and an increase in short-lived opportunistic detritivores.

Localised improvements in benthic health have been observed at sites in the Yacht Club Basin and Salamander Bay following construction of the boat dock (historically nearly all benthic fauna were eliminated from the area due to high levels of trace metals and organic contaminants being released during construction). A similar decline occurred in 2017 at the Small Bay Jetty due to organic loading from fish waste discharge. However, infaunal communities in these areas have steadily recovered and are now almost comparable to other sites in Small Bay.

Previous surveys showed a pattern of increased abundance but reduced biomass across the Bay and Lagoon, suggesting a greater contribution from smaller, opportunistic species typical of disturbed environments. Conversely, in 2026, abundance and biomass increased at Small Bay and Big Bay, potentially reflecting a greater contribution from larger-bodied taxa, although the underlying drivers remain uncertain. This was not, however, seen for Langebaan Lagoon, relative to recent peaks. While it remains difficult to fully separate natural variability from human impacts, evidence points to anthropogenic pressures contributing to reduced diversity in parts of the Bay. The 2026 analysis further identified mud content and Total Organic Carbon, together with depth and Carbon:Nitrogen ratios, as significant predictors of benthic community composition.

Long-term monitoring trends indicate increases in abundance and biomass at both Small Bay and Big Bay since monitoring began in the late 1990s. These increases may indicate some recovery in benthic community metrics following major dredging events, although the continued low diversity and prevalence of opportunistic taxa in Small Bay suggest that recovery may be incomplete. Langebaan Lagoon also exhibited positive trends in all metrics, including species richness, abundance and biomass, over the monitoring period. Although historical data for the lagoon are limited, these trends, together with the presence of sensitive taxa, may indicate a comparatively favourable ecological condition.

Other notable trends include the return of the suspension-feeding sea pen *Virgularia schultzei* to Big Bay and Langebaan Lagoon, not seen since 2004, along with an increased contribution of larger-bodied taxa to community biomass, including the tongue worm *Listriolobus capensis* and several gastropods. In Langebaan Lagoon, biomass remains relatively high despite declining from recent peaks, while the recent increase in abundance and biomass of the sand prawn *Kraussillichirus kraussi* is noteworthy. However, the factors driving these trends remain uncertain and should not necessarily be interpreted as an overall indicator of improved ecosystem health.

Areas in Small Bay with reduced water circulation continue to support impoverished infaunal communities due to the accumulation of fine sediment, organic material and trace metals, exacerbated by human activities. The continued absence of *V. schultzei* from Small Bay, where it was previously abundant, indicates reduced habitat suitability associated with these altered sedimentary and hydrodynamic conditions. Disturbances at the Liquid Petroleum Gas berth site in Big Bay following installation of the Single Point Mooring have also been associated with declines in abundance, biomass and diversity. Recent results indicate some improvement, although continued monitoring is required to confirm recovery and detect potential human impacts. Overall, the benthic system remains in a moderate but generally improving condition, with strong spatial variability, particularly in Small Bay and around the Iron Ore Terminal.

Fish and economically important invertebrates

Juvenile fish communities in Saldanha Bay and Langebaan Lagoon continue to show pronounced interannual variability, a feature of the seine net survey monitoring record since it began in the mid-1990s. In 2026, species richness remained lower than average in Small Bay and Langebaan Lagoon, with Small Bay continuing to show a long-term decline, while species richness in Big Bay was higher than in the 2025 survey. The median fish abundance in Langebaan Lagoon in 2026 was similar to the long-term median for this region, whereas the 2026 Small Bay and Big Bay medians were roughly double the long-term medians for these regions. Despite the large inter annual variability in survey estimates, juvenile fish remained more abundant within Langebaan

Lagoon and Small Bay than the more exposed Big Bay sites, emphasising the ecological importance of these sheltered habitats as nurseries for juvenile fish.

The status of white stumpnose *Rhabdosargus globiceps* remains of major concern although recruitment improved further in 2026 (to 44.8 fish/haul or 1.7 times the 2025 average and above the long-term average since 2008), as overall abundance is still low compared to historical levels, reflecting sustained recruitment overfishing at both local and national scales. Elf *Pomatomus saltatrix* recruitment has been depressed since 2018, although the 2023–2026 surveys showed a continued positive trend, with the 2026 average the highest recorded since the 2017 peak. Harders *Chelon richardsonii* abundance declined slightly in 2026, following a 2025 peak driven by several large hauls, but remained above the recent 15-year average. Harder stock assessments indicate the national population is heavily depleted and at risk of recruitment failure. Blacktail *Diplodus capensis* continue to be almost absent from surveys since 2008, with only a single individual caught in 2026, reflecting a system-wide decline. Bluntnose guitarshark *Acroteriobatus blochii* were historically abundant in the system and it is an important target in the recreational, competitive, rock and surf fishery. It is of concern that none were recorded in 2025 for the first time in the monitoring series, and only a single individual was recorded in 2026, warranting continued close attention.

This chapter includes details on the anomalous discovery of chokka squid *Loligo reynaudii* eggs in Small Bay. As this species has never been observed spawning as far west as Saldanha Bay, this finding is of interest.

The SOB fish monitoring programme provides one of the few consistent long-term datasets available to assess recruitment strength, nursery habitat use and the status of fish and elasmobranch species within the system. The data are particularly important for identifying changes in community structure and early warning signals of declines in exploited species, while also highlighting the importance of different areas of the system for surfzone fish, including threatened and endangered species. The results therefore provide an important evidence base for conservation planning, impact assessment and fisheries management. Sustained declines in several priority fishery species highlight the need for effective management and enforcement, continued monitoring, and appropriate measures to safeguard the system's nursery function and the long-term viability of its fisheries.

Birds and Seals

Saldanha Bay and Langebaan Lagoon provide extensive and varied habitat for waterbirds and seals. This includes sheltered deep-water marine habitats associated with Saldanha Bay itself, sheltered beaches, islands, and the extensive intertidal salt marshes, mud- and sandflats of the sheltered Langebaan Lagoon. The entire area is of tremendous importance in terms of the diversity and abundance of waterbird populations.

At least 56 non-passerine waterbird species commonly use the area for feeding or breeding, with eleven species breeding on the islands of Malgas, Marcus, Jutten, Schaapen and Vondeling alone. These islands support nationally important populations of Cape Gannet, Swift Tern, Kelp and Hartlaub's Gull, four species of marine cormorant and important populations of the endemic African Oystercatcher. Except for Bank Cormorants, the populations of the seabirds breeding on the islands of Saldanha Bay were on an increasing trajectory from the start of annual monitoring in the 1980s until around 2000 due to multiple factors. Since 2000, nationally important populations of African Penguin and Bank Cormorants also used to breed on the Saldanha islands, but the number of nesting pairs of these species has dwindled to a fraction (<1%) of historical

peaks. White Breasted Cormorant and Crowned Cormorant have also declined precipitously, down to 14% and 19% respectively, of their populations at the turn of the last century. The Bank Cormorant population has been declining for three decades and is currently at just 0.34% of the 1990 peak with only one breeding pair counted in the 2025–2026 breeding season. The loss of the Saldanha Bay islands as a Bank Cormorant breeding colony is imminent.

Declines can at least in part be 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) decreased breeding success as a result of a decline in the biomass of sardines nationally, and particularly along the West Coast over the last 20 years, (3) competition for food with the small pelagic fisheries (4) predation of eggs, young and fledglings by other birds and seals, and (5) the collapse of the West Coast rock lobster stock upon which Bank Cormorants feed. The 2022 closure of a 20 km radius around important seabird breeding islands to small pelagic purse seine fishing was largely in response to continuing declines of the now Critically Endangered African Penguin. Time will tell if the closure around Dassen Island (which lies within the foraging range of several species) brings any benefits to penguins and other seabirds inhabiting the Saldanha Bay islands.

Direct amelioration actions to decrease these impacts at the islands (Pelican Watch, problem seal culling) have had mixed results, with the former proving more effective than the latter. Increasing volumes of shipping traffic and associated underwater noise pose a further threat as it may negatively impact the feeding behaviour and hence survival of some sea bird species.

The recent (July 2026) outbreak of Pilchard Herpes Virus that has spread rapidly throughout the Namibian and South African pilchard stock could also have significant negative impacts on the upcoming 2026 and future seabird breeding success. The virus appears novel in the South African pilchard stock and when it was first detected in Australian waters in the late 1990's pilchard mortality rates were estimated at 60-70%. Gannet and penguin diets shifted after the Australian PHV outbreak and declines in breeding success were recorded in the breeding seasons after the outbreak. A similar response in seabirds should be anticipated within South Africa.

The Lagoon is an important area for migratory waders and terns, as well as for numerous resident waterbird species. Decreasing numbers of migrant waders utilising the Lagoon over the last 40 years reflects a global trend, which can be attributed to loss or modification of habitat, climate change, predation, pollution, hunting and other forms of human disturbance. The fact that numbers of resident waders have also declined, however, suggests that unfavourable conditions occurred in Langebaan Lagoon as a result of anthropogenic disturbance may be partly to blame. Resident wader numbers in the winter of 2022 had dropped to the lowest recorded in the 40-year count record with just 142 recorded. However, the most recent count of 1020 birds in the winter of 2025 suggests conditions may have improved. Migratory wader counts in summer also appeared to mostly stabilise at around 3 000–7 500 birds over 2015–2026. Yet drastic population declines in four species of migratory waders, including the Ruddy Turnstone, Red Knot, Grey Plover and Curlew Sandpiper have signified a downward trend in summer migratory bird numbers. Despite notable increases in 2021, the subsequent summer counts of migratory waders were unfortunately much lower and revealed that the observed "recovery"/increase was a short-lived anomaly associated with the extraordinary global COVID-19 pandemic. That being said, the sustained decline in migratory waders between about 1990 and 2010 appear to have stabilised, albeit at about a quarter of historical levels. Resident wader numbers also appear to have stabilised since 2010 at around half of the pre-1990 levels, suggesting that management measures such as zonation have been effective in halting the declines. It is highly recommended

that the status of coastal and wading bird species continues to be monitored and that these data are used to inform and assesses the efficacy of management interventions aimed at supporting recovery of the region's birds.

Cape fur seals are amongst the largest marine predators found in and around Saldanha Bay. They are opportunistic, generalist feeders that compete with seabirds, such as penguins and gannets, as well as with commercial fisheries, for small pelagic fish which form a key part of their diet. It has been suggested that the increasing numbers of seals on Vondeling Island may lead to increased pressure to cull seals both from a fisheries perspective as well as to protect important seabird species on which seals are known to prey. Concerns have also been raised that, with the increased number of seals along the shores surrounding Saldanha Bay, and should finfish aquaculture develop in the Bay, seal numbers will likely increase further along with the occurrence of problem seals. Cape Fur Seals are attracted to the aquaculture infrastructure within Saldanha Bay, in particular the rafts that they use extensively as haul outs for resting. It remains to be seen if the 2025 record of nearly 27 000 pups on Vondeling was just a fluctuation around carrying capacity or if numbers will continue to increase in coming breeding seasons. Even though the overall seal population within Southern Africa has remained stable over the last 30 years, changes in population distribution in response to environmental and ecological drivers is ongoing. Rabies was first detected within the South African seal population in June 2024. Testing of samples dating back to August 2022 revealed that rabies was already present in seals and the detection of the virus in seals from Melkbostrand to Plettenberg Bay suggests that it has been circulating in the population for some time and is now endemic. The presence of rabies in the Cape fur seal population requires vigilance and a rapid medical response should any person or their pet be bitten by a seal.

Alien and invasive species

Human-induced biological invasions are a major cause for concern worldwide. Biological invasions can negatively impact biodiversity, may result in local or even the global extinction of indigenous species, and can have tangible and quantifiable socio-economic impacts. Most of the introduced species in South Africa have been recorded in sheltered areas such as harbours/ports and are believed to have been introduced through shipping activities, largely via ballast water and hull-fouling. Given ongoing discoveries and evolving species statuses, the list of alien species in South Africa is dynamic and at the time of publication, 145 introduced marine species have been recorded, with 64 classified as invasive, 38 as cryptogenic, 37 as alien, and six as 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).

There is an increased need to implement more efficient protocols to prevent the introduction, and provide sufficient monitoring, of alien and invasive species. A precautionary approach to prevent biological invasions is often considered the most efficient method of management and can include identifying and managing important pathways of introduction. Regular monitoring is imperative to enable the early detection of alien species, guide rapid responses and support targeted control and management measures before populations proliferate, thereby reducing the impacts of these invaders on the receiving environment and biota as far as possible.

In total, 47 alien species have been recorded in Saldanha Bay and Langebaan Lagoon, of which 40 are considered invasive, and two naturalised. This alien fauna is largely comprised of the phyla Arthropoda (amphipods, crabs, barnacles), Chordata (ascidians or sea squirts), Mollusca (bivalves and gastropods), and Bryozoa (moss animals). No new alien species have been recorded in Saldanha Bay and Langebaan Lagoon since the 2025 report. However, the previously

recorded alien anemone *Cylista ornata* has recently been correctly identified as *Cereus pedunculatus*, and preliminary genetic results from the Joint Institute for Nuclear Research (Russia) confirm the presence of *Mytilus edulis*, or a hybrid of this species and *M. galloprovincialis*, in Saldanha Bay.

Continuous monitoring of the Western pea crab as part of the State of the Bay survey has been ongoing since 2004. Abundance fluctuates annually, with this species being most abundant in Big Bay, followed by Small Bay, with a potential presence in Langebaan Lagoon. The highest number of individuals was recorded in 2026 since monitoring began, with 1 520 documented in Big Bay, and 816 in Small Bay. The impact and role of the pea crab in Saldanha Bay's benthic community remains undetermined, highlighting the necessity for more in-depth studies and potential management actions.

The number of alien species recorded in the system is of concern, and highlights the need for management action, which should primarily focus on invasive species already present in Saldanha Bay. Dedicated sampling effort targeting these species should be implemented to better understand their abundance, distribution and potential impacts in Saldanha Bay and Langebaan Lagoon. For some species, there may be a need to explore control or eradication programs for effective management. Secondly, efforts should be focused on preventing further invasions. This is especially important with current world events. For example, the ongoing conflicts in the Middle East led to the closure of the Strait of Hormuz in February 2026, which may trigger a bioinvasion super-spreader event due to stranding of thousands of ships. If trade routes remain as they were before the closure, South Africa's east coast ports will be at risk from potentially new introductions within a month of ships resuming travel, but by a year after free transit, all South African ports will be at risk. Watchlists have been identified as a useful preventative measure and are created based on selecting species with an invasion history, pathways to the area of concern, occurrence in similar climatic regions, or biological traits that could predispose them to becoming successful invaders. Another vital aspect includes identifying and managing important pathways of introduction. This should be done in collaboration with Transnet Port Authority to monitor vessels entering harbours, enforce mandatory treatment of hulls and ballast water before port entry, and regularly survey harbours. All these efforts require in-depth research that will not only contribute towards our understanding of the drivers and traits governing successful invasions but also give insight into associated impacts. This could be used to support directed management actions for successfully controlling invasions and mitigating impacts.

Aquatic macrophytes

There are distinct intertidal macrophyte habitats that exist within Langebaan Lagoon: seagrass beds, such as those of the eelgrass *Zostera capensis*; salt marsh, dominated by cordgrass *Spartina maritima* and *Sarcocornia perennis*; reeds such as the common reed *Phragmites australis*, bulrush *Typha capensis*, and dune slack rush *Juncus kraussi*. Each of these habitats and their associated vegetation are important in their own right, while their interconnectivity facilitates essential ecological dynamics. They increase habitat diversity in the Lagoon, provide an important food source, stabilise sediment, offer protection to juvenile fish and invertebrates from predators, and generally support higher species richness, diversity, abundance and biomass of invertebrate fauna compared to unvegetated areas.

This chapter presents results from satellite imagery and opportunistically collected *in situ* data which assess the extent and spatial variability of seagrass, salt marsh, and common reed habitats in Langebaan Lagoon, from 2015–2026. The ground-truthing data were used to assess

annual changes in macrophyte habitat extent over the monitoring period. Common reeds were included in this analysis as it is an indicator species in the sense that its presence and health is indicative of significant groundwater inflows to the Lagoon.

Seagrass extent was found to be highly variable over time, with transient and patchy meadows present throughout the shallow areas (<2 m deep) of most of the Lagoon. The most persistent seagrass meadows were observed at Centre Banks, between Oesterwal and Shark Bay, and at Bottelary. Across these areas, 5.44 ha of seagrass was consistently detected annually. The greatest overall seagrass extent was recorded in 2019 (302 ha), followed by a general decline in extent in subsequent years. The 2026 classification, however, detected an increase in seagrass extent to approximately 227 ha, comparable to levels observed in 2023. While this may indicate renewed seagrass growth, the apparent increase should also be interpreted cautiously, as it may partly reflect improvements in classification accuracy resulting from the incorporation of opportunistically collected validation data.

Along the coastline of the Lagoon, seagrasses frequently form ecotones with coastal macrophytes, often growing in mosaics with salt marsh vegetation. This pattern is particularly evident along the coasts of Rietbaai, Klein Oesterwal to Oesterwal, Bottelary, and at the head of the Lagoon at Geelbek. Salt marsh habitat was also detected between Preekstoel and Churchhaven, at Boerplein, Schrywershoek, the head of the Lagoon at Bokramsloot, and most extensively between Geelbek and Bottelary. While core areas of salt marsh habitat varied little over the monitoring period, the extent of the habitat margins varied substantially between years, suggesting a gradual shoreward migration of the salt marsh boundary. The analysis identified 205.83 ha of persistent salt marsh habitat that remained unchanged throughout the monitoring period. Common reed cover has also changed little over the last 12 years. A large patch of persistent reed beds was detected along the eastern shoreline, extending inland from the Lagoon margin. This large, stable patch is bordered by patches of new growth and small areas where reed cover has been lost, likely indicating natural fluctuations at the edge of the distribution. 33.52 ha of stable, unchanging common reed extent was identified throughout the monitoring period.

The results highlight the importance of continued monitoring of these interconnected habitats to distinguish natural spatial and temporal variability from longer-term ecological change. Continued refinement of the classification approach, supported by targeted ground-truthing, will improve confidence in estimates of habitat extent and help identify areas requiring further investigation. Particular attention should be given to apparent changes in seagrass and common reed extents, which may provide important indicators of emerging environmental pressures within the Lagoon. Overall, the lack of any major changes in these vegetation communities points to a healthy ecosystem that is not under any significant pressure at this stage.

Management and monitoring recommendations

The current state of health of Saldanha Bay and Langebaan Lagoon has been assessed across the range of environmental parameters. In 2026, findings provide an overview of current monitoring efforts, areas of concern and identify management actions needed to address detected environmental pressures and threats. They also highlight where existing monitoring approaches may need to be adapted in response to changes in the environmental condition of the system. Based on these, an overview of current monitoring efforts, areas of concern and management actions needed to address detected environmental pressures and threats are provided. In summary, monitoring and management interventions should focus on the following issues:

Activities and discharges

- 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. Furthermore, the impact on the Saldanha Bay marine environment from projects that are primarily land-based, such as storage facilities for crude oil and LPG, 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 EIAs.
- Wastewater recycling should continue as wastewater production increases. A flow gauge monitoring the volume of wastewater entering the Bay via the Bok River should be installed.
- All discharges to the Bay, including discharges into rivers entering the marine environment, must be properly licensed and monitored (both effluent volume and quality) to confirm that conditions at the edge of the mixing zone are compliant with legislative requirements. There is evidence of a deterioration in some water quality parameters over time (such as pH for the Langebaan WWTW), and investigations should be made to ascertain and rectify the cause.
- Consideration should be given to the effects of anthropogenic ocean noise when planning or proposing projects which would increase shipping activity or underwater noise.
- Declaration of Saldanha Bay and Langebaan Lagoon as a Special Management Area in terms of ICMA should continue to be pursued.

Groundwater

- Existing and future increases in groundwater abstraction from the Langebaan Road and Elandsfontein Aquifers require careful consideration. Groundwater levels and basic water chemistry (electrical conductivity) should continue to be monitored monthly at established boreholes, with data uploaded quarterly. Current monitoring indicates stable groundwater levels and quality and a localised influence from Elandsfontein mine dewatering; continued monitoring is required to confirm this and support planned numerical groundwater-flow models and scenario testing.

Water quality

- The SBWQT monitoring of temperature in Small Bay and temperature and salinity in Langebaan Lagoon should continue but be expanded spatially and to include nutrients and pH. DFFEs dissolved oxygen monitoring in Small and Big Bay should also continue, particularly given the occurrence of severely hypoxic and anoxic bottom waters in farmed areas of Small Bay. Nitrate and other relevant water-quality data from the mariculture areas should be incorporated into the State of the Bay Monitoring Programme.
- Bacterial indicator monitoring should continue at all sites. Intestinal enterococci should be incorporated alongside, or in place of, faecal coliforms in accordance with revised South African Water Quality Guidelines and in alignment with global best practise.
- The Mussel Watch Programme should continue, with increased collaboration with trace-metal monitoring of mussels and oysters from mariculture farms. Departmental testing frequency should be increased where practicable. Results should be made publicly available and appropriate warning signage regarding the harvesting and consumption of mussels from contaminated areas should be considered.
- The Mussel Watch monitoring should be extended to include other seafood, particularly fish.

- Increasing shellfish production in Small Bay should be approached cautiously because of frequent, periodic hypoxic/anoxic conditions.

Coastal erosion

- The Saldanha Bay Municipality should continue evaluating shoreline erosion mitigation. The effectiveness of groynes and other coastal stabilisation measures, sediment accumulation and erosion in adjacent coastal compartments should continue to be assessed.
- Sea-level rise and increasing storm frequency and intensity should be incorporated into coastal vulnerability assessments.
- The remote-sensing-based coastal erosion monitoring programme should continue and be expanded through targeted monitoring of environmental drivers, including wind speed/direction, significant and extreme wave height/direction, sea-level change and individual storm events. Unmanned Aerial Vehicle surveys, beach profiling, dune and vegetation-boundary mapping and sediment assessments should be incorporated where practicable.
- CoastSnap or similar citizen-science approaches should be investigated to provide more frequent observations between satellite acquisitions and capture rapid storm-related changes.

Benthic systems

- Annual benthic monitoring should continue to track recovery and longer-term change in infaunal communities, particularly in areas affected by historical and ongoing anthropogenic pressures. Areas of restricted circulation and localised organic enrichment, such as in Small Bay, require particular attention.
- Continued standardised monitoring of the Big Bay reef epifaunal community is recommended. This monitoring is needed to detect change and distinguish potential aquaculture effects from natural variability.
- Where practicable, future finfish aquaculture within the ADZ should be directed outside mapped reef habitat, particularly in Outer Bay.

Fish and commercially important invertebrates

- Long-term seine-net surveys should continue as an early-warning system for poor recruitment and declines in important fishery species. These should be complemented by BRUV and eDNA surveys and biennial diving surveys of abalone and rock lobster.
- Recreational fishing effort and catch monitoring should be improved, particularly for white stumpnose and elf, given the importance of the recreational sector and current lack of systematic CPUE data.
- Management interventions should be considered for priority fishery declines. White stumpnose remains at historically low abundance; previously recommended reductions in the daily bag limit from 10 to 5 fish per person and an increase in the minimum size from 25 cm to 29–30 cm Tail Length should be reconsidered against the latest monitoring data.
- The recommendations of Horton et al. (2019) for reduced harder fishing effort and increased minimum mesh size should be considered to facilitate recovery. Stricter enforcement against illegal netting should also be pursued.
- The Langebaan Lagoon MPA continues to provide important protection for fishery resources. Its protective function, compliance and enforcement should be maintained, with particular attention to protecting important nursery habitats.

Birds and seals

- Coastal and wading bird populations should continue to be monitored through annual island counts and biannual CWAC surveys. The 2026 pilchard herpesvirus outbreak should be considered when interpreting future seabird breeding success and population trends, particularly where changes may reflect prey availability.
- The Pelican Watch programme should continue, with effectiveness assessed through monitoring of breeding populations and breeding success.
- Cape fur seal and Kelp Gull predation should also continue to be monitored, with management interventions considered where necessary.

Alien and invasive species

- Management of invasive species should follow a dual approach: controlling established species with known impacts while preventing new introductions.
- Taxonomic and genetic investigations used to verify unconfirmed species and improve the alien-species inventory.
- Routine surveillance and species watchlists should be maintained to support early detection and rapid eradication or control of newly introduced species. This should include targeted monitoring of high-risk pathways such as shipping and recreational vessels.
- Shipping and recreational vessels should receive particular attention as vectors for marine alien species. Hull-fouling and ballast-water controls should be strengthened, including measures addressing biofouling on international recreational yachts.
- The Draft Ballast Water Management Bill (2017) should be finalised, promulgated and implemented as a matter of urgency.

Aquatic macrophytes

- Aquatic macrophytes, particularly seagrass and salt marsh in Langebaan Lagoon, should continue to be monitored and protected. Additional physical parameters, including sediment characteristics and particle-size distribution, should be incorporated into monitoring to better understand factors influencing macrophyte condition and distribution.
- Field observations should continue to be linked to satellite mapping to improve understanding of seasonal variability and long-term change.
- The invasive bryozoan *Amathia verticillata* should be monitored, particularly around Geelbek, given its potential effects on *Zostera capensis*. Seasonal surveys should assess its extent and abundance and relationships with seagrass cover and condition.
- Macrophyte maps should be ground-truthed at key sites such as Centre Banks and Geelbek.
- Citizen-science platforms such as iNaturalist and Seagrass Spotter should be promoted as complementary sources of georeferenced observations of seagrass, epiphytes, salt marsh, reeds and invasive species.
- The potential for seagrass restoration in Langebaan Lagoon should continue to be assessed.

Socioeconomics

- The value of the Saldanha Bay and Langebaan Lagoon system to ecological, fisheries, aquaculture, tourism and recreational interests should continue to be recognised in

management planning. Monitoring results should be used within an adaptive management framework to identify emerging environmental pressures, evaluate the effectiveness of existing interventions and inform future management measures.

Closing statement

The 2026 State of the Bay report represents another valuable addition to the long-term record of environmental change in the Saldanha Bay and Langebaan Lagoon system. The continued accumulation of monitoring data is steadily improving our understanding of this complex and dynamic environment, helping to distinguish natural variability from human-induced change and providing an increasingly robust evidence base for management.

While important challenges remain, the monitoring programme continues to highlight areas of improvement, emerging pressures and opportunities for more effective management. As the programme enters its third decade, the continued commitment to long-term monitoring and collaboration between stakeholders will remain critical to understanding and protecting this unique system. We look forward to the 20th State of the Bay report, which will mark an important milestone in this ongoing record of the environmental health and changing character of Saldanha Bay and Langebaan Lagoon.

Langebaan Lagoon

1 INTRODUCTION

1.1 BACKGROUND

Saldanha Bay and the adjacent tidal Langebaan Lagoon, situated on the west coast of South Africa approximately 100 km north of Cape Town, are areas of exceptional beauty and South African biodiversity "hot spots". Indeed, a number of Marine Protected Areas (MPAs) have been proclaimed in and around the Bay and Langebaan Lagoon, with much of the surrounding land protected as part of the West Coast National Park (Figure 1.1). Langebaan Lagoon and the Saldanha Bay islands (Schaapen, Marcus, Malgas, Jutten and Vondeling) are also declared Ramsar sites, because of their importance to waterbird and seabird populations.

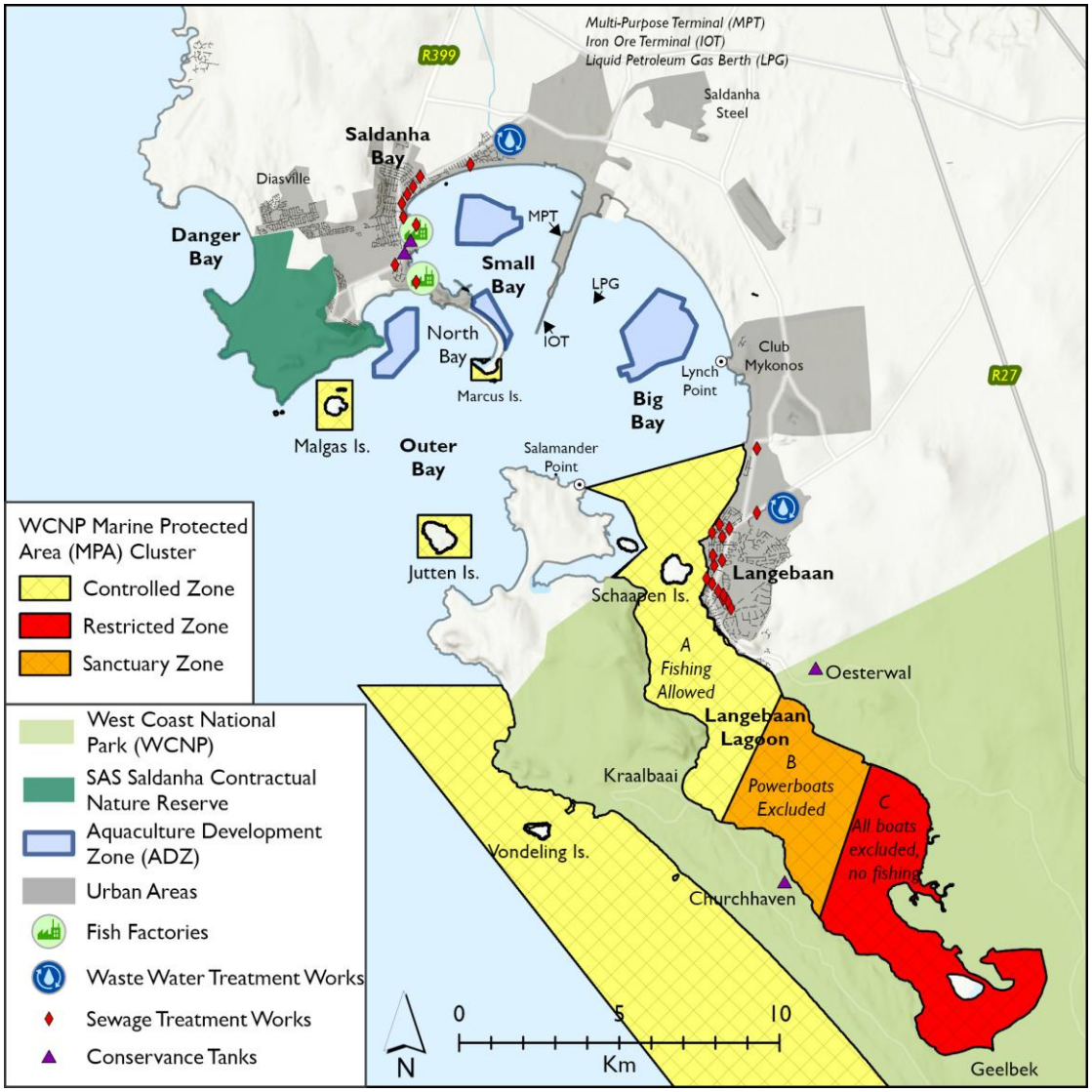

Figure 1.1. Regional map of Saldanha Bay, Langebaan Lagoon and Danger Bay showing development (grey shading) and conservation areas.

The conservation importance and unique positioning and features of Saldanha Bay and Langebaan Lagoon have resulted in a long history of scientific study and interest. 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 to draw together information from the various studies focused on the area. This 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. In the intervening years, there has been expansive development within, and changes to, the form and function of the Saldanha Bay system, including the construction of an iron ore export facility in the 1970s and the Multi-Purpose Terminal in the 1990s. This 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.

The need to promote an integrated approach to the management, conservation and development of the waters of Saldanha Bay and the Langebaan Lagoon, as well as the adjacent land areas, in the light of both historical and ongoing development culminated in the inauguration of the Saldanha Bay Water Quality Trust (SBWQT) in 1996. This voluntary organization, renamed as the Saldanha Bay Water Quality Trust (WQT), represents various organs of State, local industry and other relevant stakeholders and interest groups. It 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.

The WQT recognised that regular, long-term environmental monitoring is essential to identify and enable adaptive management and mitigation of negative human impacts on the environment, and in so doing, maintain the beneficial value of an area for all users. As such, it initiated the monitoring of important ecosystem indicators in 1999 which cumulated in the “State of the Bay” (SOB) report series, beginning in 2006, and which has been produced annually since 2008. The report series serves 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. Monitoring of water quality, sediment quality and benthic macrofauna has been undertaken since 1999, with monitoring of surf zone fish and rocky intertidal macrofauna added to the programme in 2005. Several additional indicators have been added to the monitoring programme over the years, including a summary of the activities and discharges that affect the health of the system, and an overview of management and policy development as it pertains to Saldanha Bay and Langebaan Lagoon. Other additions include a discussion on the groundwater resources of the surrounding areas, an assessment of shoreline erosion in Saldanha Bay and surrounding areas, and an overview of known and newly recorded alien and invasive species present in the system. Given the results of more than 15 years of annual monitoring, and in accordance with the new approved 10-year monitoring plan for the WQT, some parameters (including sediment quality, aquatic macrophytes and rocky intertidal communities) are now monitored and reported on every second year rather than on an annual basis.

This 2026 report is the 19th in the series and provides an update on the health of all monitored parameters in Saldanha Bay, Langebaan Lagoon and Danger Bay. This report

presents the 2026 data in the context of work commissioned by the WQT, historical trends pre-dating the State of the Bay series, a range of other scientific monitoring programmes and studies, as well as trends reported on in previous reports (2006, 2008–2025).

Despite the noteworthy successes in environmental monitoring and the conservation of important marine habitats within the West Coast National Park Marine Protected Area (MPA), the historical and on-going development of the area has resulted in environmental degradation of some marine and coastal habitats. There are cases where management intervention has resulted in significant improvements in ecosystem health, whilst ongoing efforts should be made to rehabilitate existing damage and mitigate against the potential environmental impacts of future development.

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 thirteen chapters, each of which addresses different aspects of the health of the system.

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

Chapter Two provides background information to anthropogenic impacts on the environment and the various approaches used to monitor these impacts, with a specific focus on capturing the differences in the nature and temporal and spatial scale of these impacts.

Chapter Three provides a comprehensive overview of the national, provincial and local environmental management measures and legislation that govern the greater Saldanha Bay area.

Chapter Four 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 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 and status of the Bay.

Chapter Seven assesses recent and seasonal patterns of coastal erosion and accretion along the beaches of Saldanha Bay and Langebaan Lagoon, examines the influence of

wave conditions on shoreline movement, and provides recommendations for improved monitoring and coastal management.

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

Chapter Nine summarises all available information on the fish community and composition in the Bay and Lagoon, and presents results from a surf zone fish monitoring programme using research seine nets that was initiated in 2005.

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

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

Chapter Twelve provides an overview of aquatic macrophytes present in Langebaan Lagoon and the pressures affecting these habitats. The chapter also presents opportunistically collected ground-truthing data used to refine the results of a supervised classification effort, which assessed the annual changes in habitat extent over the period of 2015–2026.

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

1.3 WHAT'S NEW IN THE 2026 EDITION

Readers who are familiar with the State of the Bay (SOB) report series will know that while the various chapters of this report are updated each year with new data and information that has been collected during the course of the preceding twelve months, either through focused surveys commissioned by the WQT 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 may be a little tedious to wade through for those who have seen it all before. This section of the report therefore serves to highlight the new data and information included in each of the data 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: Coastal and environmental policy and management

Since 2024 several important legislative updates were introduced, and new and revised regulations, guidelines, and policies were also published under various Acts dealing with climate change, air quality, coastal management, waste, water, marine spatial planning, marine resources and land use. In 2025/26, key changes include the introduction of the 2026 National Water Amendment Bill to the National Assembly in January 2026, and the introduction of the Water Services Amendment Bill (B24–2025) to the National Assembly

in October 2025. An updated national Biodiversity Assessment (NBA) was also released in 2025.

At the local level, several key management and planning instruments for the Saldanha Bay and Langebaan area have been updated since publication of the 2025 State of the Bay report. The Saldanha Bay Municipality adopted the fourth annual review of its 5th Generation Integrated Development Plan, which covers the period 2022–2027, and approved the 2025–2030 Municipal Spatial Development Framework in May 2025. The Western Cape Land Use Planning Amendment Bill [B 3–2025] progressed through the Western Cape Provincial Parliament, with the Premier assenting to the updated law as the Western Cape Land Use Planning Amendment Act, 2026 (Act No. 2 of 2026) in July 2026. The State of Knowledge and Situation Assessment Report has been completed for the West Coast National Park (Kock et al. 2025), along with the stakeholder processes for the draft West Coast National Park Marine Protected Areas, Islands and Langebaan estuarine lagoon management plan (Alison Kock, *pers. comm*. 2026). The coastal management line for West Coast National Park is also currently under revision (Alison Kock, *pers. comm*. 2026).

Chapter 4: Activities and discharges affecting the health of the Bay

Only developments and activities which have experienced changes since the last SOB report are retained in this chapter. There were a few changes to the structure and layout of this chapter compared to previous years. Furthermore, over the years there have been a number of changes in how specific data are recorded or reported. This year changes have been implemented to correct some of the reporting problems picked up from various sources. These include correcting the assumptions of vessels data from Transnet National Ports Authority (TNPA), including both ballast water discharge and vessel numbers. Also, tourism numbers drawn from StatsSA are now 'overnight' tourists and exclude 'day-visitors' which were previously counted as tourism visits.

Updated information included in the sections of this chapter is listed below:

- Numbers of visitors to the West Coast National Park (WCNP);
- Metal exports from the Saldanha Bay Multi-Purpose and Iron Ore Terminals;
- Information on new and existing development proposals for Saldanha;
- Shipping traffic and ballast water discharges;
- Effluent discharges into Saldanha Bay, including 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; and
- Mariculture industry in Saldanha Bay, including a summary of the results of the latest ecological monitoring survey.

Chapter 5: Groundwater

Groundwater continues to play a vital role in sustaining the ecosystems of Saldanha Bay and Langebaan Lagoon, with stable groundwater levels and electrical conductivity trends indicating no negative impacts from current wellfield abstraction and other groundwater activities in the area. The municipal monitoring network remains at 24 boreholes, and updated wellfield abstraction volumes, groundwater levels and electrical conductivity data for the past year have again been included; abstraction volumes were notably lower

this year following vandalism at both wellfields in September 2025. For the first time, the chapter also includes a dedicated assessment of the Elandsfontein mine's groundwater monitoring cross-section from upgradient of the mine to the Langebaan Lagoon, confirming that the mine's dewatering-and-recharge system is keeping its influence localised around the pit, with no evidence of impact reaching Langebaan Lagoon.

Chapter 6: Water quality

This chapter presents some new information on variations in temperature and salinity, at various locations in the Bay and at the head of the Lagoon, microbial indicators relating to bathing safety, as well as trace metals in mussels on the shoreline and offshore mariculture facilities. Arsenic concentrations inshore-based and farm-based bivalves were often above guideline limits for almost all samples collected in 2026, which is a concern.

Microbial indicators (*E.coli*) revealed encouraging results at most sites in 2026 for recreational use limits, with most sites showing Excellent water quality (19 of 20 sites, the best year on record). The Lagoon bathing beaches were rated 'Excellent' in 2026 including Langebaan Main Beach which was rated as Poor last year. Hoedjiesbaai maintained its Excellent status from 2025, however, Bok River Mouth again remained the most contaminated in terms of microbial concentrations. The relatively new faecal coliforms Water Quality Guidelines for mariculture, that are more stringent than the previous guidelines (DWAF 1995a), meant most sites were non-compliant, from both Small Bay (as expected) and even Big Bay. Small Bay sites, Hoedjiesbaai, the beach at the caravan site and Bok River mouth were worst offenders. Historically, Big Bay has generally complied with mariculture water quality guidelines due to effective tidal flushing and lower point source pollutant inputs. In 2026, however, several sites showed a broader decline in microbiological water quality, characterised by persistently, but only slightly, elevated faecal coliform concentrations rather than isolated pollution spikes. This suggests a temporary increase in background faecal contamination, potentially associated with diffuse land-based runoff, stormwater inputs, sewer infrastructure issues, or other catchment-scale influences.

Finally, water quality monitoring activities that were initiated as part of the Environmental Monitoring Programme (EMPr) for the Aquaculture Development Zone are providing valuable new insights into oxygen dynamics in the Bay and have again highlighted the impacts of reduced water circulation and organic loading in Small Bay.

Chapter 7: Coastal erosion

The 2026 edition further expands the coastal monitoring programme by examining shoreline change at a seasonal scale and statistically assesses its relationship with significant wave height. Shoreline delineations were also improved using locally observed water levels and transect-specific beach slopes, providing a more accurate representation of shoreline movement. The assessment confirms continued erosion between the Langebaan groynes, substantial accretion south of these structures, and a moderate-strong association between increased wave height and erosion along Langebaan North and central Paradise Beach. It also introduces event-based monitoring and proposes a CoastSnap community science initiative to help capture rapid shoreline changes that may be missed by satellite imagery.

Chapter 8: Benthic systems

This chapter provides new insights into the species composition, abundance, biomass and health of benthic infauna communities in Saldanha Bay and Langebaan Lagoon, based on the 2026 sampling survey. It also summarises past changes in sediment composition (particle size distribution) and quality (trace metal, organic carbon, and nitrogen content) over time and in relation to dredging events, with a focus on the implications for macrofaunal communities.

Chapter 9: Fish and commercially important invertebrates

This chapter presents new results from the 2026 seine net surveys of juvenile fish. Fish species richness remained lower than average in Small Bay and Langebaan Lagoon, with Small Bay showing a clear long-term decline, while Big Bay richness improved on the previous year. White stumpnose recruitment has increased in recent years but remains low, elf numbers have risen since 2023 to their highest since 2017, and harders remain overexploited despite their abundance. Blacktail remain almost absent and bluntnose guitarshark, which were not recorded at all in 2025 for the first time in over two decades, remained scarce with only a single individual recorded in 2026. This year, chokka squid eggs were observed for the first time in Small Bay, and this unusual occurrence is described.

Chapter 10: Birds and seals

This chapter provides an update on the species threat status and presents the most recent available data on the composition, abundance and health of birds breeding and feeding in Langebaan Lagoon and on the islands in Saldanha Bay. It also presents data on trends in the biomass of small pelagic fish (sardines and anchovies) that are the most important prey species for many of the endangered sea birds that breed on the Saldanha Bay islands. The 2022 small pelagic fishery closure around Dassen Island (which lies within the foraging range of several species) does not appear to have resulted in significant increases in the numbers of penguins utilising the islands in Saldanha Bay as the number of breeding pairs continues to decrease. However, the drivers of penguin population decline are numerous and improving food availability is not the panacea for all of these. The number of breeding Cape Gannets on the Malgas Island has increased modestly in recent years, which may be partly a result of improved foraging efficiency associated with the purse seine fishery closures but also the small increases in sardine biomass along the west coast in the last four years. Unfortunately, a Pilchard Herpes Virus (PHV) outbreak recently (July–September 2026) recently spread through the South African and Namibian pilchard population and this could compromise any recovery of stocks with negative trophic impacts on seabirds. The presence of rabies in Cape fur seals appears to be endemic and still poses a water user safety issue with numerous attacks on people and dogs reported from around the coast.

Chapter 11: Alien and invasive species

This chapter provides an updated account of alien and invasive species based on the latest surveys and developments since the 2025 report. Currently, 145 introduced marine species have been recorded from South Africa, with 64 classified as invasive, 38 as cryptogenic, 37 as alien, and six as 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). Key updates for 2026 include: 1) Forty-seven alien species are now confirmed from Saldanha Bay and Langebaan Lagoon, of which 40 are considered invasive, and two naturalised. No new alien species have been reported since 2025; 2) Additional eDNA data was collected during the 2025 and 2026 SOB surveys as part of a program funded by Anglo Shipping, in collaboration with NatureMetrics; 3) Updated quantitative data on the Western pea crab *Rathbunixa occidentalis* is provided, with the highest number of individuals recorded in 2026 (since monitoring began in 2004); 4) The previously recorded alien anemone *Cylista ornata* has recently been correctly identified as the naturalised European daisy anemone *Cereus pedunculatus* and; 5) Preliminary genetic results from the Joint Institute for Nuclear Research (Russia) confirm the presence of *Mytilus edulis*, or a hybrid of this species and *M. galloprovincialis*, in Saldanha Bay.

Chapter 12: Aquatic macrophytes

The new additions to this chapter describe how the 2026 macrophyte monitoring programme used Sentinel-2 satellite imagery in conjunction with opportunistically collected ground-truthing data to assess the extent and spatial variability of seagrass, salt marsh, and common reed habitats in Langebaan Lagoon from 2015–2026. The ground-truthing data were used to refine a supervised classification approach, enabling the assessment of annual changes in macrophyte habitat extent over the monitoring period. Additionally, opportunistic vegetation samples were collected and brought to a laboratory, where the various epiphytes found to be growing on these macrophytes were observed under dissecting microscopes and identified to the lowest taxonomic level possible.

Chapter 13: Management and monitoring recommendations

This chapter provides an updated summary of the key changes detected in each parameter covered in this report and assigns a health status rank to each. It also includes updated recommendations on management actions that need to be implemented to mitigate key threats highlighted in the previous chapters.

Mussel rafts in Big Bay

2 BACKGROUND TO ENVIRONMENTAL MONITORING

2.1 INTRODUCTION

In South African law, pollution is defined by the National Environmental Management Act (No. 107 of 1998) (NEMA) as amended as '...any change in the environment caused by (i) substances ... (iii) noise, odours, dust or heat emitted from any activity, including the storage or treatment of waste or substances, construction and the provision of services, whether engaged in by any person or an organ of state, where that change has an adverse effect on human health or well-being or on the composition, resilience and productivity of natural or managed ecosystems, or on materials useful to people, or will have such an effect in the future'.

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. Trace metals refer to certain metals that are naturally found in very low concentrations in the environment which, in higher concentrations, can be highly toxic to aquatic organisms. These include mercury, cadmium, arsenic, lead, chromium, zinc and copper. These metals occur naturally in the earth's crust, but mining of metals is increasing the rate at which these are being mobilised. 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 oil (such as 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 pose a particular risk to sea birds. Seabirds are particularly vulnerable to being coated by spilled oil, and chronic and acute oil pollution is a significant threat to both pelagic and inshore species (Vanstreels et al. 2023). Oiled seabirds may be more vulnerable to hypothermia, as oil reduces their insulation. Organochlorines do not occur naturally in the environment and are manufactured entirely by man, and include 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 may cause eutrophication of coastal and inland waters. Eutrophication in turn can result in proliferation of algae, phytoplankton blooms, and deoxygenation of the water resulting in hydrogen sulphide production (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 concentrations of pollutants, and their effects on the aquatic/marine environment can be detected through the analysis of pollutant concentrations 1) in the water itself, 2) in sediments or 3) in living organisms. Each of these approaches has advantages and disadvantages, and a comprehensive monitoring programme typically requires that a variety of parameters be monitored covering water, sediment and biota.

The analysis of pollutant concentrations in water samples may be challenging because these pollutants can be difficult to detect without highly sophisticated sampling and analytical techniques, even at concentrations lethal to living organisms. Pollutant concentrations in natural waters may vary season, state of the tide, currents, extent of freshwater runoff, sampling depth, and the intermittent flow of industrial effluents, which complicates matters even further. To accurately elucidate the degree of contamination of a particular environment, many water samples usually need to be collected and analysed over a long period of time. The biological availability of pollutants in water also presents a problem in itself—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).

Analysis of pollutant concentrations in sediments 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 leads to pollutants becoming concentrated in sediments over time. By analysing concentrations in the sediments 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 detection is generally easier, since concentrations tend to be much higher in sediment than water.

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. 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 (Phillips 1980, Phillips & Rainbow 1993). 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.

To overcome this, contaminants can also be assessed 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 integrate concentrations over time and

can reflect concentrations in the environment over periods of days, weeks, or months depending on the type of organism selected. Different types of organisms also 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, 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). Commonly selected organisms for bio monitoring include bivalves (e.g., mussels and oysters) and algae (i.e., seaweed). However, not all pollutants accumulate in the tissues of living organisms, including, for example, nutrients and particulate organic matter. Therefore, while it is advantageous to monitor contaminant concentrations in biota, simultaneous monitoring of sediment and water quality is also necessary.

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 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 therefore indicate a change in environmental conditions. Monitoring abundance or biomass of a range of different organisms from different environments and taxonomic groups with different longevities (invertebrates, fish and birds, for example), offers the most comprehensive perspective on change in environmental health spanning months, years and decades.

2.3 INDICATORS OF ENVIRONMENTAL HEALTH AND STATUS

The environmental parameters and indicators selected for assessment provide an overview of the broader environmental health and status of Saldanha Bay and Langebaan Lagoon. Indicators were selected based on their relevance to key environmental processes and pressures, as well as the feasibility of measuring changes over appropriate spatial and temporal scales. Parameters or indices presented in this report are described below:

- Certain activities (including shipping and small vessel traffic, the mere presence of people and their pets as well as trampling) can cause disturbance in the environment, especially to sensitive species. This disturbance, 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), increases in the abundance of pest species (e.g., harmful algal blooms), or the introduction of alien species. Monitoring these patterns and levels of these activities can provide insight into the reasons for any observed changes to/deterioration of in

ecosystem health and can help in formulating solutions to address negative trends.

- 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.
- Coastal development includes infrastructure construction and operation (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 difficult to assess fully by project specific impact assessments. These 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.
- Structures erected in the coastal zone such as harbours, breakwaters and buildings (as well as dredging activities), can also 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 and coastal protection infrastructure has been installed.
- Benthic macrofauna are mostly short-lived organisms (1–3 years) and are therefore 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.
- Fish are mostly longer-lived animals (3–10+ years) that 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 are also mostly long-lived animals (6–15+ years) that are good indicators of long-term changes in the health of the environment. They are particularly susceptible to hydrocarbon pollution, disturbance by human presence, shipping activities (underwater noise),infrastructural development (e.g., housing development), and changes in food supply (e.g., small pelagic fish and intertidal invertebrates).
- South Africa has at least 145 confirmed alien species, most of which are considered invasive. With new introductions and discoveries every year, this list remains dynamic with several additions underway. Most alien 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. As ballast water tends to be loaded in harbours, alien species often originate from these

sheltered habitats. Although they have difficulty adapting to the more exposed coastline, they easily gain a foothold in sheltered bays such as Saldanha Bay.

2.4 ECOSYSTEM HEALTH SCORES

A ranking system has been devised for the Saldanha Bay and Langebaan Lagoon State of the Bay monitoring programme that incorporates both the drivers of changes (such as 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 for a range of different organisms) (Table 2.1). Collectively, these parameters provide a comprehensive picture of the State of the Bay and 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.

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 continue and management interventions to facilitate recovery.
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.

Cormorants in the Small Bay ADZ precinct

3 MANAGEMENT AND POLICY DEVELOPMENT

3.1 SPATIAL CONTEXT

Rapid urban and industrial growth presents significant threats to the natural habitat and ecological integrity of marine and coastal areas in Saldanha Bay and Langebaan Lagoon, especially given the nationally and internationally important ecosystems and habitats delineated in the area. For example, the Saldanha Bay and Langebaan Lagoon system lies within an Ecologically or Biologically Significant Area (EBSA), defined by the Convention on Biological Diversity (CBD) as "geographically or oceanographically discrete areas that provide important services to one or more species/populations of an ecosystem or to the ecosystem as a whole, compared to other surrounding areas or areas of similar ecological characteristics...". The principal objective of EBSAs is the identification of features of higher ecological value that may require enhanced conservation and management measures. The Saldanha Bay-Langebaan Lagoon system falls within the Cape Canyon and Associated Islands, Bays and Lagoon EBSA, which is considered critical for ecosystem functioning and species survival (MARISMA 2020). EBSAs are also important for identifying key marine biodiversity sites and in guiding future Marine Spatial Planning (MSP), particularly the establishment of Marine Protected Areas (MPAs) (Kirkman et al. 2021, Harris et al. 2022).

Langebaan Lagoon, Sixteen Mile Beach, Malgas Island, Jutten Island, and Marcus Island are all classified as MPAs and collectively fall within the West Coast National Park (WCNP) (Figure 3.1). Surrounding terrestrial areas include several protected areas, the largest being the WCNP to the south, and four nature reserves to the north (Figure 3.1).

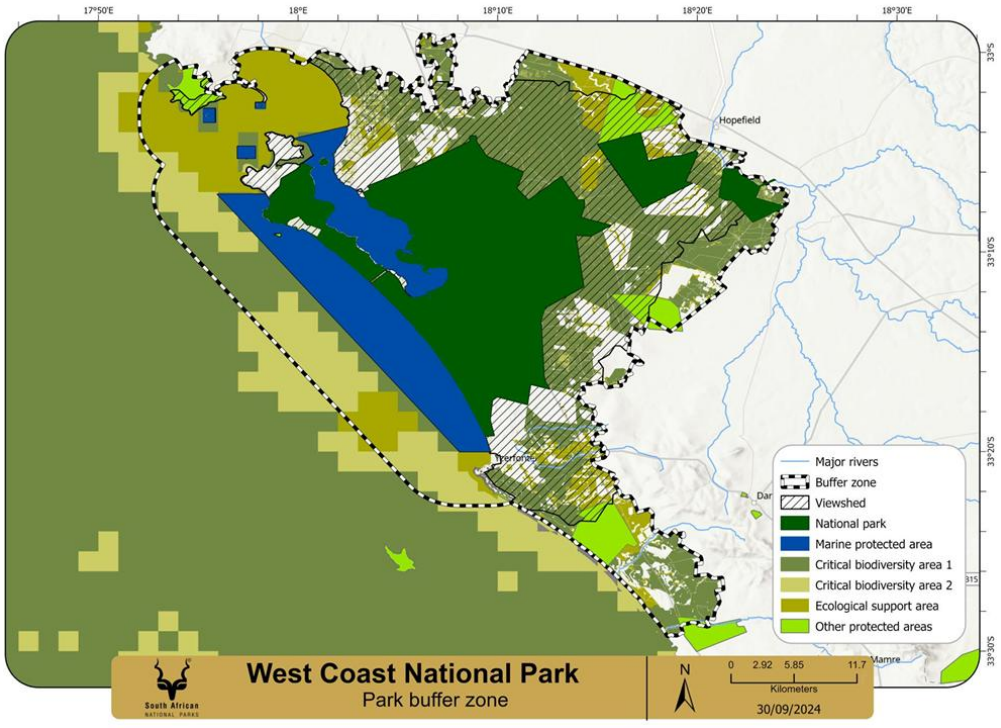

Figure 3.1. Terrestrial and Marine Critical Biodiversity Areas in the Saldanha Bay area. (SANParks 2025).

The WCNP protects a rich diversity of plant and animal life characteristic of the West Coast, provides critical habitats for both endemic and migratory species, supports ecotourism, and helps maintain the long-term ecological health of the system (SANParks 2025). The WCNP also forms the core conservation area of the Cape West Coast Biosphere Reserve, recognised by the UNESCO Man and the Biosphere Programme. Langebaan Lagoon and the offshore islands (Schaapen, Marcus, Malgas, and Jutten Islands) are also designated as Ramsar Wetlands of International Importance for their ecological value and role in supporting biodiversity and livelihoods (Ramsar 2024).

Informed by the importance of these areas, the 2024 Western Cape Biodiversity Spatial Plan (BSP) for terrestrial and marine Critical Biodiversity Areas (CBAs) (CapeNature 2024, Marine Biodiversity Sector Plan 2024) classify most of the Bay as an Ecological Support Area (ESA), with CBA Restore zones near the mouth of the Bay and Lagoon (Figure 3.1). Terrestrial areas along the coastline consist of a mosaic of CBA 1 and ESA areas, with the largest portion being CBA 1, indicating a largely natural or near-natural state.

These classifications make the area a high conservation priority, where development is restricted or must be carefully mitigated to prevent further ecosystem loss. Management plans address these priorities by guiding conservation, land use, and sustainable development, while supporting compatible activities like tourism and recreation.

Although many developments occurring within the area are land-based, increased vessel traffic associated with these activities adds to the volume of ballast water discharged into the Bay (see Chapter 4: Activities and Discharges). Wastewater and stormwater volumes are also rising annually, contributing cumulatively to environmental pressures. Mussel and oyster aquaculture, together with wild-capture fisheries, also contribute to the cumulative impacts on the marine environment (Chapter 4: Activities and Discharges). These activities, while economically significant, can affect water quality, benthic habitats, and local nutrient dynamics, particularly in areas where operations are concentrated. The expansion and intensification of aquaculture in particular may interact with other stressors such as pollution, climate change, and invasive species, requiring integrated and adaptive management to balance ecological sustainability with economic development in the Bay.

The current and desired future state of the greater Saldanha Bay area is complex, with industrial development, including the Saldanha Bay Industrial Development Zone (IDZ), directly adjacent to conservation zones such as Ramsar sites, MPAs, and National Parks. The area supports multiple, potentially conflicting, uses including industry, fisheries, mariculture, recreation, and natural ecosystems. This requires sustainable development that directs growth towards transformed, resilient locations while protecting sensitive areas (Thérivel et al. 1999).

Understanding management practices in the Bay requires familiarity with the applicable legislation. This chapter outlines the key national, provincial, and local legislation and planning frameworks governing the area (Section 3.3) and lists some of the key updates and amendments to legislation that are relevant to the area (Section 3.4).

3.2 LEGAL FRAMEWORK

South Africa's environmental protection is regulated at national, provincial, and local levels. The Constitution of the Republic of South Africa, 1996, is the supreme law and provides the fundamental legal basis for environmental management. It guarantees the right to an environment that is not harmful to health or well-being and requires its protection for present and future generations through appropriate legislative and other measures.

National legislation includes primary laws (Acts), subordinate legislation (Proclamations, Regulations, Norms and Standards, and Lists of Activities), and Guidelines that support implementation. These are administered by departments such as the Department of Forestry, Fisheries and the Environment (DFFE), the Department of Water and Sanitation (DWS), or the South African Heritage Resources Agency (SAHRA). The National Environmental Management Act (NEMA; Act 107 of 1998) is the overarching framework for environmental protection and sustainable development. It is supported by Specific Environmental Management Acts (SEMAs), including the NEM: Protected Areas Act (NEMPAA; Act 57 of 2003), NEM: Biodiversity Act (NEMBA; Act 10 of 2004), NEM: Air Quality Act (NEMAQA; Act 39 of 2004), NEM: Integrated Coastal Management Act (ICMA; Act 24 of 2008), and NEM: Waste Act (NEMWA; Act 59 of 2008). These are further supported by various other key environmental laws. Provincial legislation supports national laws and addresses province-specific issues such as pollution control, conservation, and regional planning. In the Western Cape, the Department of Environmental Affairs and Development Planning (DEA&DP) is the lead authority.

At the local level, municipalities such as the Saldanha Bay Municipality (SBM), develop their own policies and by-laws to regulate land use, zoning, environmental management, and infrastructure, which align with broader provincial and national objectives. They are required to prepare spatial plans and frameworks, guided by national and provincial legislation and informed by higher-level spatial and biodiversity data such as the Western Cape BSP and the recently released 2025 National Biodiversity Assessment (NBA). Examples of key legislation and instruments prepared and applied by municipalities include:

- Zoning Scheme and Planning By-Laws: Regulates land use rights, zoning and consent use applications.
- By-Laws: Regulates management of stormwater, water use, air quality, and waste.
- Integrated Development Plan (IDP): A primary strategic plan which guides municipal development priorities, including budgeting, development, service delivery, safety, socio-economic growth, and environmental sustainability. Ensures alignment with national and provincial legislation and plans.
- Municipal Spatial Development Framework (SDF): Provides spatial direction for municipal growth and development while safeguarding sensitive environments.
- Municipal Environmental Management Framework (EMF): Identifies environmental sensitivities, defines compatible and incompatible activities, and provides guidance to prevent environmental degradation.

3.3 KEY LEGISLATION

An overview of the key national and provincial legislation governing the Saldanha Bay and Langebaan Lagoon area is provided below (Table 3.1). The most relevant local spatial planning instruments, and management and monitoring plans, programmes, and frameworks, along with any recent updates that apply specifically to the Saldanha Bay and Langebaan Lagoon system are also summarised (Table 3.1). These are drafted as required by the relevant National and Provincial legislation and guide development, conservation, and resource use in the area. This list is not exhaustive.

Table 3.1. Key national and provincial legislation, international agreements/ conventions and local plans, programmes and frameworks of relevance to Saldanha Bay and Langebaan Lagoon.

Legislation/ Plan/ Framework Name	Description
International Agreements and Obligations	
Conventions, programmes, summits and organisations	• World Heritage Convention, 1972: Convention Concerning the Protection of the World Cultural and Natural Heritage • Convention on International Trade in Endangered Species of Wild Fauna and Flora, 1973 • Bonn Convention, 1979: Convention of Migratory Species of Wild Animals (1979) • Abidjan Convention, 1981: The Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region • United Nations Convention on Biological Diversity (CBD), 1992 • United Nations Framework Convention on Climate Change, 1992 • Global Programme of Action for the Protection of the Marine Environment from Land-based Activities, 1995 • Agenda 21, 1992—Confirmed at the United Nations World Summit on Sustainable Development, Johannesburg Summit, 2002 • The Sustainable Development Summit, 2015 • Ramsar Convention: Convention on Wetlands of International Importance, 1971 • United Nations Educational, Scientific and Cultural Organization (UNESCO)
National Legislation	
National Environmental Management Act (Act 107 of 1998)	NEMA provides for cooperative environmental governance by establishes principles for decision-making on matters affecting the environment. It requires identification, assessment, and authorisation of activities that may affect the environment, socio-economic conditions, or cultural heritage. Key provisions include: • Environmental Impact Assessment (EIA) Regulations and Specialist Study Protocols: Assessing impacts of certain developments with listed activities and enabling stakeholder engagement. • EMFs: Spatial tools guiding sustainable development in sensitive areas (e.g. Saldanha Bay EMF). • Norms and Standards: Activity-specific requirements (e.g. waste storage, land remediation, biodiversity offsets). • Appeals and Compliance: Appeals, enforcement, and rectifying unlawful activities. • Land Use Planning Guidance

Legislation/ Plan/ Framework Name	Description
	• Strategic Environmental Assessment (SEA) and Environmental Management Instruments (EMIs): Tools for integrating environmental and sustainability considerations into policy and planning. • Biodiversity and Ecosystem Guidelines: Promotes inclusion of ecosystem services and biodiversity priorities in planning and decision-making.
National Environmental Management: Protected Areas Act (Act 57 of 2003)	Provides for the formal declaration and management of Protected Areas (e.g. National Parks, Nature Reserves, MPAs) to conserve biodiversity, protect natural and cultural resources, and maintain ecological integrity. Requires management plans to guide conservation and land-use decisions and assigns management responsibility to designated authorities (e.g. SANParks).
The National Environmental Management: Biodiversity Act (Act 10 of 2004).	Governs biodiversity management and the protection of threatened species and ecosystems through regulations and permits. It gives effect to international agreements (e.g., the CBD, 1992), supports planning tools such as bioregional and Biodiversity Management Plans, regulates alien and invasive species, and establishes the South African National Biodiversity Institute (SANBI) for research, monitoring, and reporting. The Act also mandates biodiversity assessments like the NBA, which informs conservation and planning.
National Biodiversity Framework	A national-level strategic policy document mandated under Section 38 of the NEMBA. It provides a coordinated framework to guide the management, conservation, and sustainable use of South Africa's biodiversity, setting priorities, targets, and strategies across all sectors and spheres of government. The framework is reviewed and updated every five years
Animals Protection Act (Act 71 of 1962)	Consolidates and amends laws relating to the prevention of cruelty to animals.
National Veld and Forest Fire Act (Act No. 101 of 1998)	Governs the prevention and combating of veld, forest, and other fires, and provides for fire protection associations and fire management responsibilities in South Africa.
National Environmental Management: Air Quality Act (Act 39 of 2004)	Provides the framework for regulating air quality in South Africa. It establishes Air Quality Management Plans, requires licences for listed activities that produce atmospheric emissions, sets National Ambient Air Quality Standards for priority pollutants, and allows for the declaration of priority areas where air quality is degraded.
National Environmental Management: Integrated Coastal Management Act (Act 24 of 2008)	Provides for integrated coastal and estuarine management to promote conservation and ensure that coastal development is socially, economically, and ecologically sustainable. It defines rights and duties, regulates activities that may harm the coastal environment, and gives effect to international coastal obligations. It also guides the establishment of coastal management lines and development and implementation of management plans, programmes and protocols. It further regulates effluent discharges into the coastal environment.
National Coastal Management Programme	Provides the framework for implementing integrated coastal management at national and provincial levels.
National Estuarine Management Protocol	Provides guidance for managing estuaries and developing Estuary Management Plans (EMPs), including integration into coastal and protected area management frameworks.
Marine Pollution Act (Prevention of Pollution from Ships; Act 2 of 1986)	Provides for the protection of the sea from pollution by oil and other harmful substances discharged from ships and gives effect to the International Convention for the Prevention of Pollution from Ships, 1973, as amended by the Protocol of 1978.
The Marine Living Resources Act (MLRA; Act 18 of 1998)	Provides for the sustainable management of marine living resources and promotes the protection of entire ecosystems and marine biodiversity. It enables the declaration of MPAs where certain activities may be restricted to

Legislation/ Plan/ Framework Name	Description
	support conservation goals. It also regulates commercial, subsistence, and recreational fishing.
Sea Fisheries Act (Act 12 of 1988)	Largely repealed, with only Sections 1, 29, 38, 47, 48, and 50 remaining in force to govern levies, collection of aquatic plants and shells, offences, and related enforcement measures.
Sea Birds And Seals Protection Act (Act 46 of 1973)	Prohibits the harassment, disturbance, killing, or capture of seabirds and seals. It is supported by the South African Policy on the Management of Seals, Seabirds, and Shorebirds, 2007, aimed at reducing incidental mortality caused by fishing operations.
National Environmental Management: Waste Act (Act 59 of 2008)	Provides a framework for sustainable waste management to protect human health and the environment by reducing resource use and waste, promoting reuse and recycling, and ensuring safe disposal. It establishes a National Waste Management Strategy, sets norms and standards, requires licences for waste activities, regulates hazardous waste, mandates local waste planning and reporting, and covers contaminated land remediation.
Hazardous Substances Act (Act 15 of 1973)	Provides for the classification, control, and regulation of hazardous substances and certain electronic products that may pose risks to human health or safety, including restrictions on their import, manufacture, sale, use, and disposal.
National Water Act (Act 36 of 1998)	Provides the framework for the protection, use, development, conservation, and management of water resources. One of the main provisions under this Act is the requirement to obtain permission for certain water uses through either a Water Use Licence (WUL) or a General Authorisation (GA).
Water Services Act (Act 108 of 1997)	Regulates the provision of water services, including water supply and sanitation, and establishes the framework for the rights and duties of water services authorities and providers.
Marine Spatial Planning Act (Act 16 of 2018)	The Act establishes a national system for MSP to manage ocean use across sectors, promote sustainable development, conserve marine ecosystems, and resolve conflicts between competing uses. It is based on the 2017 National MSP Framework and guides the preparation of Marine Area Plans.
Spatial Planning and Land Use Management Act (Act 16 of 2013)	Provides a national framework for spatial planning and land use decisions to ensure coordinated, sustainable, and equitable development. It guides zoning, aligns planning with other legislation, and empowers municipalities to prepare SDFs and land use systems that integrate environmental, social, and economic objectives.
Special Economic Zones Act (Act 16 of 2014)	Governs the establishment, development, and management of Special Economic Zones (SEZs).
National Heritage Resources Act (NHRA; Act 25 of 1999)	Sets out the framework for protecting and managing heritage resources, which includes cultural, archaeological (including shipwrecks), and palaeontological resources. It also establishes the South African Heritage Resources Agency (SAHRA) and provincial heritage authorities to provide oversight.
Mineral And Petroleum Resources Development Act (Act 28 of 2002)	Makes provision for equitable access to and sustainable development of the nation's mineral and petroleum resources. Requires the need for prospecting and mining rights and permits.
Gas Act (Act 48 of 2001)	Provides for the orderly development of the piped gas industry through a national regulatory framework, with the National Gas Regulator as its custodian and enforcer.
Electricity Regulation Act (Act 4 of 2006)	Establishes a national framework for regulating South Africa's electricity supply industry. It designates the National Energy Regulator as the authority responsible for oversight and enforcement. The Act governs licensing and registration for electricity generation, transmission, distribution, trading, and import/export, including municipal reticulation.

Legislation/ Plan/ Framework Name	Description
Carbon Tax Act (Act 15 of 2019)	Imposes a tax on the CO_2 equivalent of greenhouse gas emissions to encourage emission reductions, with supporting regulations on tax rates, reporting, and allowances, and amendments to align with evolving climate policies.
The Climate Change Act (Act 22 of 2024)	Facilitates an effective climate change response and support a long-term, just transition toward a low-carbon, climate-resilient economy and society in South Africa, aligned with principles of sustainable development.
Infrastructure Development Act (Act 23 of 2014)	Facilitates and coordinates priority public infrastructure projects of economic or social importance, promoting state development goals and improving management across all phases from planning to operation.
Local Government: Municipal Systems Act (Act 32 of 2000)	Provides the framework for developmental local government, promoting social and economic upliftment, universal access to affordable services, community participation, effective municipal administration, and coordinated planning, performance management, and service delivery.
Promotion of Access to Information Act (PAIA; Act 2 of 2000)	Gives effect to the constitutional right of access to information held by the state or by another person when required to exercise or protect rights and assigns related powers and duties to the Information Regulator.
Promotion of Administrative Justice Act (Act 3 of 2000)	Gives effect to the constitutional right to lawful, reasonable and procedurally fair administrative action, including the right to written reasons for such action.
Disaster Management Act (Act No. 57 of 2002)	Establishes a legal framework for disaster risk management, including the establishment of the National Disaster Management Centre and the development of disaster management frameworks at national, provincial, and local levels.
Provincial Legislation	
Western Cape Biodiversity Act (Act 6 of 2021)	Provides the provincial framework for conserving, protecting, and sustainably managing biodiversity and ecosystems. It recognises and mandates the use of the BSP as a key tool in planning and regulatory processes and integrates biodiversity considerations into land-use planning and decision-making. Some sections have been in effect since 15 November 2022, while others will be phased in through the development of Regulations. Once fully operational, the Act will repeal older legislation consolidating biodiversity management into a single statute.
Western Cape Provincial Biodiversity Spatial Plan	A provincial biodiversity planning tool that identifies priority and sensitive areas, classifying them as CBAs, ESAs, Protected Areas or Other Natural Areas based on ecosystems, species, and ecological processes. Data are presented in CBA Maps. It is used to integrate biodiversity into planning and decision-making, guiding conservation and sustainable development.
Nature and Environmental Conservation Ordinance (Ordinance 19 of 1974)	Governs nature conservation in the Western Cape, including protection of wild flora and fauna. Once the WCBA is fully in effect, this Ordinance will be repealed. Until such time, relevant administrative and law enforcement provisions remain governed by the Ordinance.
Western Cape Biosphere Reserves Act (Act 6 of 2011)	Provides for the designation and management of biosphere reserves in the Western Cape. It regulates land use within these reserves through the preparation of framework plans and addresses related incidental matters.
Western Cape Land Use Planning Act (Act 3 of 2014)	Governs spatial planning and land use management in the province. It guides the drafting of provincial and municipal SDFs and ensures that these align with national priorities and the Spatial Planning and Land Use Management Act. It ensures that infrastructure, heritage, agriculture, tourism, ecologically important areas, biodiversity, soil conservation, pollution control, and climate change adaptation considerations are incorporated into planning. It also guides applications such as rezoning, subdivision, and consent use.
The Western Cape Provincial Spatial	The province's highest-level statutory spatial plan, prepared and adopted under the Western Cape Land Use Planning Act. It sets the spatial vision and

Legislation/ Plan/ Framework Name	Description
Development Framework	development principles for the province and ensures that all municipal SDFs align with provincial and national spatial and land-use priorities.
Western Cape Provincial Coastal Management Programme	A strategic framework guiding sustainable coastal management, conservation, and planning. It is mandated under ICMA, which requires provinces to implement coastal management programmes aligned with national policies.
Local Spatial Planning, Management and Monitoring Instruments	
West Coast National Park Management Plan	The WCNP includes Langebaan Lagoon and its surrounding terrestrial and coastal habitats. The park is managed by South African National Parks (SANParks) under the NEMPAA. The public participation processes for the draft West Coast National Park Marine Stakeholder Protected Areas, Islands and Langebaan estuarine lagoon management plan has been completed (Alison Kock, *pers. comm*. 2026).
Marine Protected Area Management Plans/ Estuary Management Plan	The WCNP is zoned to regulate recreational and commercial activities, with some areas allowing swimming, angling, and boating, and others reserved for strict conservation. The offshore islands provide critical breeding and roosting habitat for seabirds such as Cape Gannet, Kelp Gull, and Cape Cormorant (many of which are Red Listed). Terrestrial areas protect habitats threatened by agriculture and housing development. Langebaan Lagoon and the offshore islands (Schaapen, Marcus, Malgas, and Jutten Islands) are designated as a Ramsar Wetlands for their ecological value and role in supporting biodiversity and livelihoods (Ramsar 2024). Ecologically, it serves as a key spawning and nursery ground for several endemic fish species, supports diverse invertebrate and seaweed communities, and conserves about 32% of South Africa's saltmarshes. The Lagoon is also listed under the Bonn Convention as an important non-breeding site for hundreds of thousands of Palaearctic migrant waders during the austral summer (Summers 1977). Where a national park and MPA or estuary overlap, NEMPAA requires integrated management, while ICMA requires that Estuary Management Plan (EMPs, mandated for all estuaries) be incorporated into the park's management plan when the estuary falls within its boundaries. Ramsar obligations further require the Lagoon's management to be included in the park's planning framework. SANParks has therefore developed an integrated plan for the WCNP MPAs, encompassing the Lagoon, offshore islands, and terrestrial areas, that fulfils the requirements of an MPA management plan, an EMP, and Ramsar commitments.
Saldanha Bay Municipality Integrated Development Plan	The fourth annual review of the 5th Generation IDP for 2026/27 was undertaken. This updated IDP covers the period 2022–2027 and incorporates new socio-economic data, community development strategies, and capital investment planning, and is aligned with Spatial Planning and Land Use Management Act principles of equity, sustainability, and good governance (Saldanha Bay Municipality 2025).
Municipal Spatial Development Frameworks	The 2025–2030 Municipal SDF was formally approved by Council on 29 May 2025 (Provincial Gazette No. 9089, 20 June 2025) (Saldanha Bay Municipality 2025). Once the new Provincial SDF is adopted, the Municipal SDF will be updated to ensure alignment.
Environmental Management Framework	First reviewed in the context of the Greater Saldanha Regional Spatial Implementation Framework in 2018 (DEA&DP 2018). It was subsequently updated and published in its current form in 2021, with its scope expanded to include the Berg River and estuary (DEA&DP 2021). No further updates have been published.
Coastal Management Programme	Reviewed and updated in 2019 for the period 2019–2025 (SBM 2019). It sets out ten coastal management objectives to be achieved through defined strategies, with progress monitored and measured against specified indicators. No further update has been published.
Strategic Environmental	The most recent SEA for the Greater Saldanha Bay area was completed in 2019 (DEA&DP 2019) and identified significant risks to natural capital. It noted that

Legislation/ Plan/ Framework Name	Description
Assessments for the Greater Saldanha Bay Area	marine biodiversity and ecosystems were approaching critical thresholds due to habitat loss and industrial activities such as oil and gas development. No subsequent updates have been published.
Saldanha Strategic Offset Strategy, 2020	Drafted in 2020 to manage potential conflict between CBAs and the area designated for large scale industrial expansion around the Port of Saldanha and the Saldanha Bay IDZ.

3.4 RECENT UPDATES

Recent key updates and amendments to national and provincial legislation and subordinate legislation that are relevant to the area are outlined (Table 3.2). Updates to spatial plans, frameworks, and SBM-specific management plans under these Acts are presented in Table 3.1 above.

Table 3.2. Summary of recent key updates and amendments to National and Provincial Legislation and Subordinate Legislation in 2025–2026.

Act	Updates
Amendments to Legislation	
Electricity Regulation Amendment Act (Act 38 of 2024)	Came into effect on 1 January 2025. Updates the 2006 Electricity Regulation Act by revising licensing rules, enabling new generation capacity, creating the Transmission System Operator SOC Ltd, and introducing a competitive open electricity market.
New Regulations/ Guidelines published under Acts	
The National Environmental Management: Biodiversity Act (Act 10 of 2004).	The updated NBA was released in November 2025.
National Environmental Management: Air Quality Act (Act 39 of 2004)	National dust control regulations were gazetted for March 2026, and the previous 2013 National Dust Control Regulations have been repealed. The purpose of these regulations is to prescribe measures for the control of dust in all areas nationally and steps that must be taken to prevent nuisance by dust. The 2026 Dust Control Regulations provide strong penalties for any person convicted of an offence under the regulations.
Integrated Coastal Management Act (Act 24 of 2008)	The revised draft of the National Coastal Management Programme of South Africa, 2025–2030, has been published in the Government Gazette no. 52388 of 28 March 2025. The coastal management line for West Coast National Park is currently under revision (Alison Kock, *pers. comm.* 2026). The public participation processes for the draft West Coast National Park Marine Stakeholder Protected Areas, Islands and Langebaan estuarine lagoon management plan have also been completed (Alison Kock, *pers. comm.* 2026).
National Environmental Management: Protected Areas Act, 2003 (Act No. 57 of 2003)	A State of Knowledge and Situation Assessment Report has been completed for the West Coast National Park as part of the development of the West Coast National Park Marine Protected Areas, Islands and Langebaan estuarine lagoon management plan (Kock et al. 2025). The public participation processes for the draft West Coast National Park Marine Stakeholder Protected Areas, Islands and Langebaan estuarine lagoon management plan have also been completed (Alison Kock, *pers. comm.* 2026).

Act	Updates
Marine Living Resources Act (Act 18 of 1998)	New Permit Conditions for the 2026 season have been published for 1) the Pelagic Fish Sardine/ Pilchard Fishery, and 2) Commercial Small nets: Gillnet/Driftnet/Setnet Fishery Area G: Langebaan-West Coast National Park.
Western Cape Land Use Planning Act (Act 3 of 2014)	Western Cape Land Use Planning Amendment Bill [B 3–2025] updates provincial land use legislation. It has progressed through the Western Cape Provincial Parliament, with the Premier assenting to the updated law as the Western Cape Land Use Planning Amendment Act, 2016 (Act No. 2 of 2026) in July 2026.
New/ Pending Laws and bills not yet in force	
National Water Act (Act 36 of 1998)	The 2026 National Water Amendment Bill (NWA Amendment Bill) was introduced to the National Assembly on 21 January 2026. The public comments deadline was 30 July 2026. Key proposed changes include: 1. Trading is prohibited, with surrendered entitlements reverting to the Minister for reallocation in the public interest. 2. Unused or underutilised water allocations may be reduced or reallocated to historically disadvantaged individuals or priority users. 3. Section 33 is to be repealed, with no new Existing Lawful Water Uses declarations. Historical users must formalise licences or risk losing access. 4. A specific volume of water must be set aside in every water management area to redress past racial and gender discrimination (proposed section 27(3)). 5. Responsible authorities must consider the applicant's compliance history, prior convictions, and prior licence revocations before issuing a licence. 6. The Minister may gazette Strategic Water Source Areas (SWSAs) and restrict or ban activities inside them, including opencast and underground mining, forestry plantations, and certain agricultural activities. 7. The Minister can reallocate water between provinces, sectors and catchments in the public interest (proposed section 25A), providing a stronger national tool for drought response and equity. 8. Stronger penalties and meaningful deterrents for large polluters and illegal abstractors. 9. Directors of companies and municipal managers may be held personally liable if they failed to take all reasonable steps to prevent an offence (proposed section 156A). Changes to representation at Water User Associations (WUAs).
Water Services Act (Act 108 of 1997)	Water Services Amendment Bill (B24–2025) was introduced to parliament on 03 October 2025. Key amendments include: 1. Providing the Minister with enforcement and rectification powers in addition to monitoring and intervention powers if water services providers do not meet their obligations effectively. 2. The proposed amendment of Section 22 of the Water Services Act lists additional criteria municipalities (as water services authorities) must apply before approving a water service provider's right to operate. 3. Also proposed is the addition of Section 22A to the Water Services Act, which prohibits any person from operating as a water service provider without an operating licence granted by DWS. 4. The proposed Section 22D allows the DWS to intervene if a water services provider fails to meet the conditions of its operating licence and, if there is persistent failure on the part of the water services provider, the proposed amendment gives DWS power to revoke the operating licence.

Act	Updates
	Overall, the proposed amendment bolsters National government's ability to hold municipalities, water boards, and water services providers accountable for underperformance.
National Heritage Resources Act (NHRA; Act 25 of 1999)	Notice of commencement to review this was put out in August 2025.
Municipal Fiscal Powers and Functions Amendment Act (Act 4 of 2024; GG 50798)	Amends Spatial Planning and Land Use Management Act to add fiscal provisions, allowing municipalities to levy development charges, adopt policies, and manage engineering service contributions. This new Act was assented to in 2024 but has not yet commenced.
National Environmental Management Act, 1998 (Act No. 107 of 1998)	Proposed regulations for the exploration and production of onshore petroleum resources requiring fracturing technology published for public comment on 7 November 2025.
Upstream Petroleum Resources Development Act (Act 23 of 2024)	Declares State sovereignty over petroleum resources, promotes equitable access, local employment, sustainable resource management, and infrastructure access. Assented to on 29 October 2024 but not yet commenced.
Draft National Environmental Management: Biodiversity Bill, 2024	Published for public comment on 24 May 2024 (GN 4887 in GG 50706). Seeks to replace NEMBA, introduce a revised regulatory approach, provide stronger protection and more flexible management of species and ecosystems, introduce additional measures for species and ecosystem listing, support the implementation of international agreements, curb wildlife trafficking, and promote economic benefits from the biodiversity sector.
Draft Aquaculture Development Bill, 2023	Published for public comment on 17 November 2023 (GN 4098 in GG 49736). It aims to promote responsible aquaculture, coordinate research, ensure effective regulation, support sector transformation, attract investment and create jobs.

Langebaan in 2016

4 ACTIVITIES AND DISCHARGES

4.1 INTRODUCTION

Saldanha grew from a small fishing village into a town that supports multiple industries largely because it is a sheltered bay along an otherwise exposed coastline. Industrial development of Saldanha Bay dates back to the early 1900s with the establishment of commercial fishing and rock lobster industry. The development of a large-scale industrial port in Saldanha Bay commenced with the construction of an iron ore export facility in the 1970s to facilitate the export of iron ore from the Northern Cape. Aerial photographs taken in 1960, 1989, 2007 and 2023 clearly show the extent of development that has taken place within Saldanha Bay over the last 50 years (Figure 4.1). The most significant developments have been the causeway linking Marcus Island to the mainland, to provide shelter for ore-carriers, and the construction of the Iron Ore Terminal (IOT). The construction of the IOT essentially divided Saldanha Bay in two, into 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 4.1). By the end of the 1970s, Saldanha Bay harbour was an international port able to accommodate large ore-carriers.

Development of the causeway and IOT in Saldanha Bay greatly modified the natural water circulation and current patterns and bathymetry in the Bay (Henrico & Bezuidenhout 2022) (see Chapter 6: Water Quality). 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. This Chapter details the historical and ongoing developments and impacts thereof within and around Saldanha Bay and Langebaan Lagoon. This includes the development of Port facilities (Section 4.3.1), and the export of metal ores (including iron, lead, copper, zinc, and manganese) (Section 4.3.2). Activities directly associated with the operation of the Port include air quality management (Section 4.3.3) and dredging (Section 4.3.4) and these are followed by shipping related activities (Section 4.4) such as shipping volume (Section 4.4.1), ballast discharges (Section 4.4.2) and shipping noise (Section 4.4.3).

Saldanha Bay has long been recognised as a strategically important industrial centre in the Western Cape, and therefore this Chapter outlines the establishment and ongoing progress of the Saldanha Bay IDZ (Section 4.4.5). There are a number of current and future energy developments under the IDZ umbrella, including the development of a Green Hydrogen hub, the development of liquid petroleum gas (LPG) import and Gas to Power facilities (Section 4.5.2).

In addition to the Port operations (and associated impacts), this Chapter also explores the socioeconomics (such as population growth and tourism) of Saldanha Bay and Langebaan Lagoon (Section 4.2). A number of these developments result in effluent discharge to the Bay, the management and impacts of which are presented in Section 4.10. The status of the Aquaculture Development Zone (ADZ) is also provided in Section 4.12.

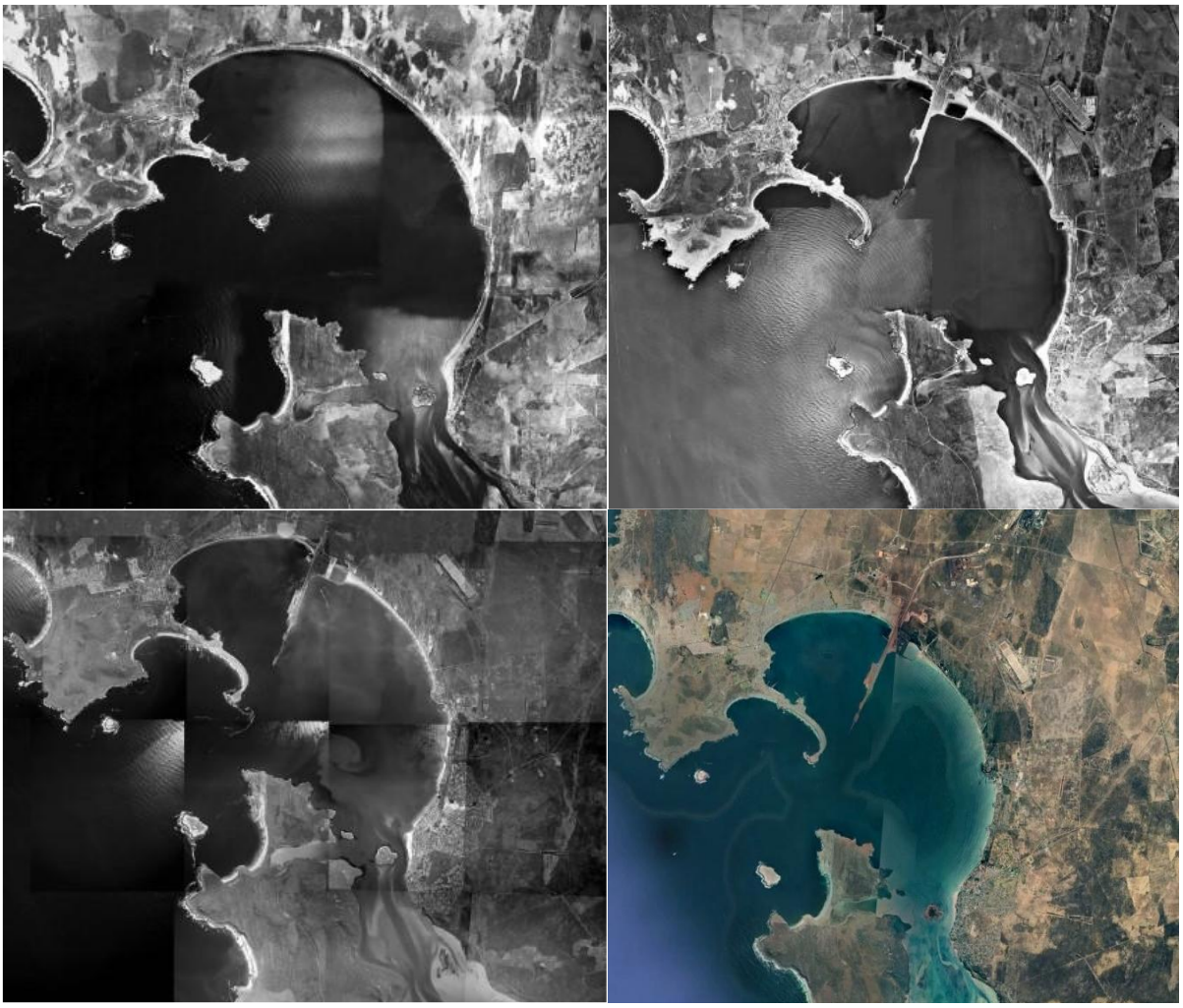

Figure 4.1. Composite aerial photos of Saldanha Bay and Langebaan Lagoon. (Top left): 1960, note the absence of the ore terminal and causeway and limited development. (Top right): 1989; note the presence of the ore terminal, the causeway linking Marcus Island with the mainland, and expansion of settlements at Saldanha and Langebaan. (Bottom left): 2007; note the expansion in residential settlements particularly around the town of Langebaan (Source: Department of Surveys and Mapping). (Bottom right): 2023, note the expansion in residential settlements particularly around the town of Langebaan and between Langebaan and the Iron Ore Terminal. Source: Google Earth 2023.

4.2 SOCIO-ECONOMICS

4.2.1 POPULATION GROWTH AND URBAN DEVELOPMENT

In 2023, Saldanha Bay's Gross Domestic Product by Region (GDPR) was valued at R9.3 billion (real basic prices), maintaining its position as the largest economy in the West Coast District (contributing 31.8% of district GDPR) and providing employment to approximately 44 856 individuals (with 38 765 engaged in formal employment). Following the 2021 post-pandemic recovery (6%), local economic growth has moderated, weighed down by national macroeconomic challenges, load shedding, and port logistics constraints. The municipal economy continues to be driven primarily by manufacturing (20.3%), finance, real estate & business services (17%), and wholesale & retail trade (12.8%). However, net job creation has lagged behind output performance, with

persistent employment pressures in primary and labor-intensive sectors such as agriculture and retail (Western Cape Government 2023/2024).

According to recent provincial economic data, the three largest sectoral contributors to GDPR in the Saldanha Bay Local Municipality are manufacturing (20.3%), finance, insurance, real estate & business services (17%), and wholesale & retail trade, catering & accommodation (12.8%), with agriculture, forestry & fishing contributing 12.6% (Western Cape Government 2023/2024).

Data from Stats SA Census 2022 indicates that the Saldanha Bay municipal area accounted for approximately 25.2% of the West Coast District Municipality's (WCDMs) total population, recorded at 124 166 individuals in 2022 and projected to reach approximately 138 500 by 2028 under revised provincial demography models (Stats SA 2023; Western Cape Government 2023). At a town level, historic growth trends demonstrate substantial urban expansion: Saldanha's population increased from 16 820 in 1996 to 28 135 in 2011, reaching over 32 000 by 2022, while Langebaan experienced marked growth from 2 735 in 1996 to 8 294 in 2011 and surpassing 14 000 in 2022 (Stats SA 2012, 2023). As of 2022, the demographic profile showed a balanced gender distribution of 49.6% females to 50.4% males (Stats SA 2023). Analysis of age cohorts reveals that while the working-age population (15–64 years) comprises the largest segment (~67.5%), the elderly category (65+ years) continues to exhibit the highest relative annual growth rate (>2.5%), reflecting both increases in overall life expectancy and the continued appeal of coastal settlements like Langebaan as retirement destinations (Western Cape Government 2023/2024).

Urban development around Langebaan Lagoon has progressively encroached upon the coastal margin, severely reducing or eliminating critical coastal buffer zones (Figure 4.2). Permitting high-density urban infrastructure to extend directly to the water's edge subjects sensitive littoral environments to acute anthropogenic stressors, including direct habitat degradation, trampling, and loss of indigenous coastal dune vegetation. This spatial footprint compromises natural coastal processes—specifically sand transport, dune migration, and shoreline equilibrium—while elevating coastal erosion risk. Furthermore, the expansion of hard, impervious surfaces increases urban stormwater discharge into the lagoon ecosystem, accelerating the transport of terrestrial contaminants, trace metals, and excess nutrient loads into the marine environment (See Section 4.10).

Figure 4.2. Satellite image of Saldanha (Small Bay, left) showing little or no set-back zone between the town and the Bay, and of Langebaan (Right) showing absence of development set-back zone between the town and the Lagoon (Google Earth 2023).

4.2.2 TOURISM

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 WCNP, 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 (2005–2025) on numbers of visitors to the WCNP indicate strong seasonal trends in numbers of people entering the park, peaking in the summer months, particularly the December holiday period, and during the flower season in August and September (Figure 4.3). Paying day guests (excluding international visitors) contribute the most to this seasonal pattern, followed by free guests[1] which peak in September due to the Annual SANParks Week. Wild Card holders (memberships) similarly peak in the flower season in September. Conversely, international guests and overnight guest numbers are relatively constant throughout the year but make up lower numbers than the other visitor categories (Figure 4.3).

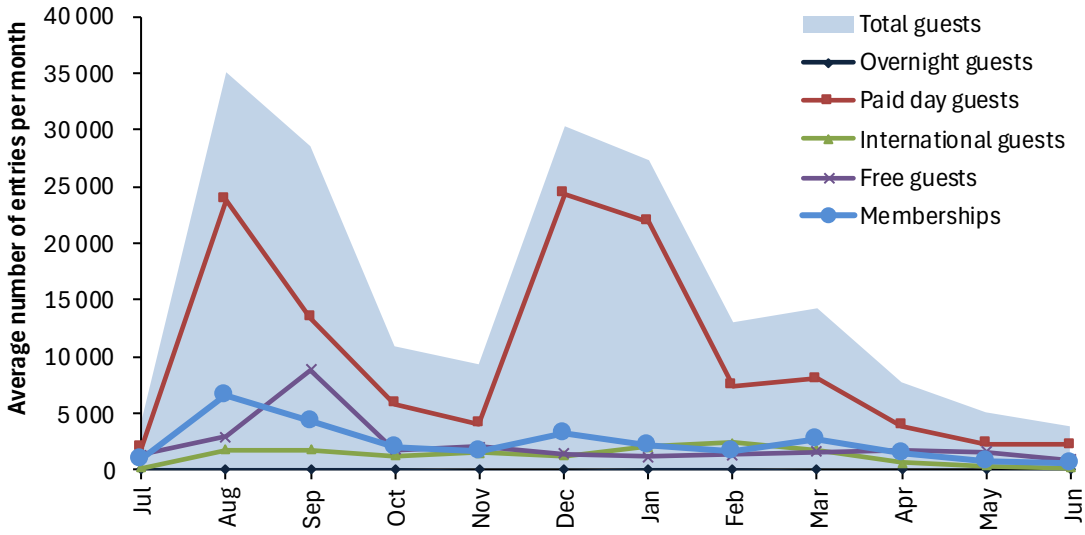

Figure 4.3. Monthly average numbers of entries into the West Coast National Park. Paid day guests include all South African visitors (adults and children), while Overnight guests refer to those staying in SANParks accommodation. International guests include all SADC and non-African day visitors (adults and children) while the category 'Free guests' includes residents, staff, military, school visits. Memberships = Wild Card holders. (Source: Pierre Nel, WCNP, 2026).

The number of free guests has increased steadily over time, and between 2013/14 and 2017/18 was similar to the proportion of day guests. However, the recent data format shows that this includes guests with memberships, such as Wild Card holders, showing that paid guests are still the most numerous category (Figure 4.4). For the 2025/2026 reporting cycle park visitor numbers were only reported as "Camping", "Accommodation", "Guest" and "Activity" visitors. To fit with previous data we lumped

[1] These include school groups, military personnel, official visits, SANParks and Concessionaire staff, residents and service provider entries.

Camping and accommodation visitors under "Overnight guests" and Guests and Activity visitors under "paid day guest" although it was not clear if these may include some proportion of free visitors.

The popularity of overnight stays inside the park decreased substantially after 2009, reaching an all-time low in the 2020/21 rolling year (12 overnight guests) and has been erratic since then. However, the last year has shown increased popularity of overnight stays with numbers reaching the highest it's been in eight years (Figure 4.4). Total number of guests and Wild Card holder numbers were only available from 2018/19 until 2024/2025. It should be noted that SANParks tourism data are now managed by the national head office and the reporting structure has been standardised across all national parks.

In 2024/25, international visitor numbers to the WCNP were similar to those of the previous year, and almost at 2010/11 levels (Figure 4.4). Since the significant drops in 2020 and 2021 due to the COVID-19 lockdowns, overall numbers of international tourists to South Africa, increased more than three-fold in 2022, increased again in 2023, with another slight increase to 2024 (Figure 4.5). Last year, 2025, was a record year for tourist visits and the first year surpassing pre-pandemic numbers.

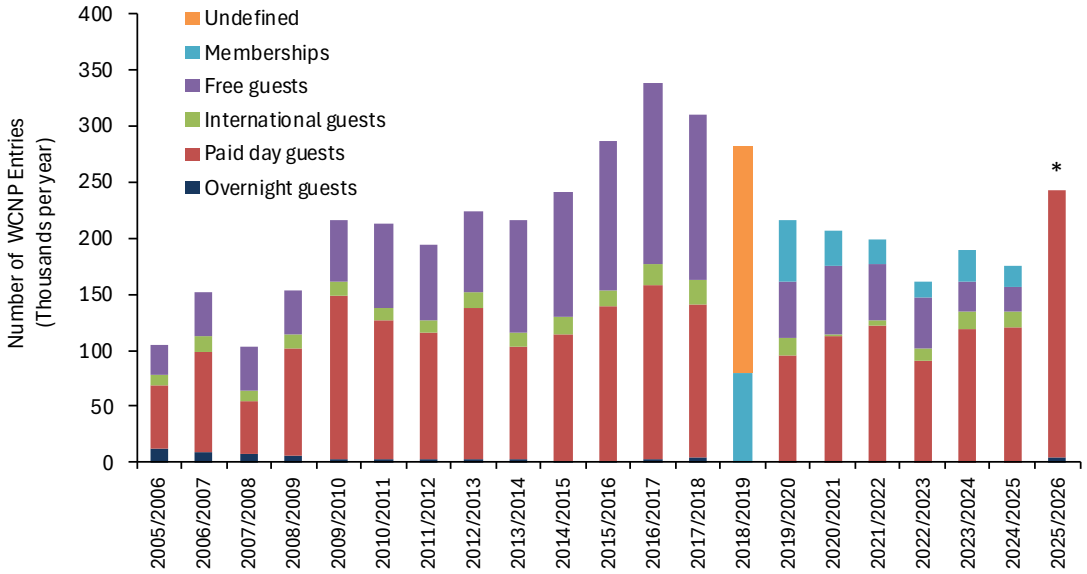

Figure 4.4. Numbers of entries into the West Coast National Park (see Figure 4.3 for additional detail). In 2018/2019 only total number of guests and wild card holders (Memberships) were recorded, however, this has changed with recent data again being categorised. The 2019–2021 'Free guests' are now divided into 'Complimentary' and 'Memberships' (Wild Cards). * Note that the 2025/2026 records only show overnight stays and daily/activity guests, without any indication on payments or memberships (Source: Pierre Nel, WCNP, 2026).

SANParks are in the process of undertaking a Basic Assessment Process for proposed upgrades of the Geelbek Precinct within the West Coast National Park. The existing Geelbek Precinct includes a tourism facility which consists of a temporary marquee tent, a restaurant, an education facility, parking, access roads and administration offices. In

November 2025 the upgraded entrance gate on the R27 (Weskus-road) was opened after being closed since April 2025 for the upgrades.

SANParks are also in the process of obtaining Environmental Authorisation for overnight facilities at Klein Mooimaak and the upgrade of Langebaan Gate. The draft Basic Assessment report for this proposed development was published by EnviroWorks, an independent Environmental Assessment Practitioner (EAP) in July 2025 (EnviroWorks 2025).

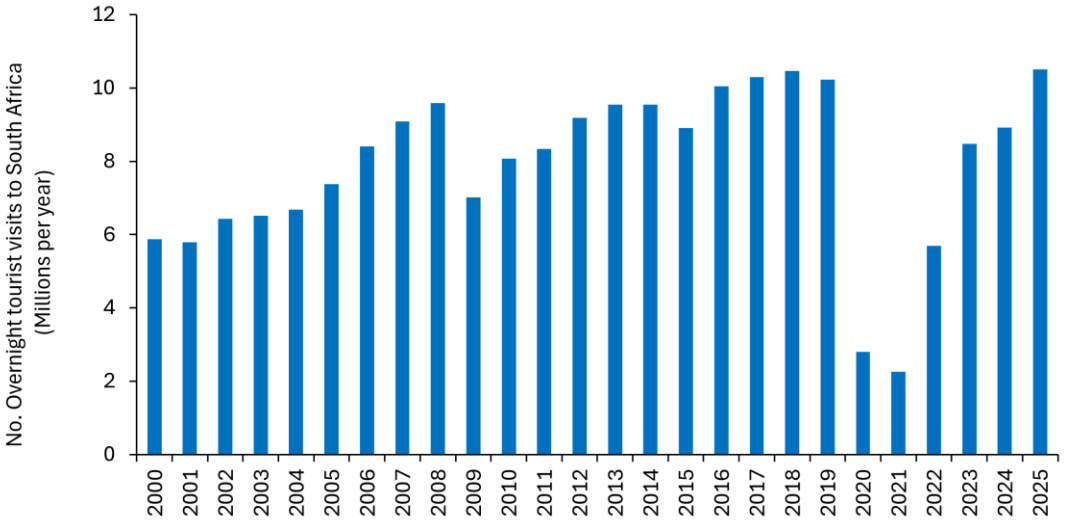

Figure 4.5. Total annual numbers of international overnight tourist visits to South Africa (Stats SA 2025). The 2025 year was the first year since the COVID-19 pandemic that reached and exceeded pre-pandemic levels with 10,5 million tourist visits.

4.3 PORT FACILITIES

4.3.1 CURRENT LAYOUT AND FUTURE PLANS

Port facilities in Saldanha Bay include the main Transnet IOT with berths for three ore carriers, an oil jetty, a Multi-Purpose Terminal (MPT) (completed in the late 1990s), and a general maintenance quay, a fishing harbour which is administered by the DFFE, 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.

Future plans for the development of the Port are shown in Figure 4.6 (TNPA 2022). In the short term, changes include (1) the amendment of Port Limits to include newly acquired land in compliance with the Terminal Operator Agreement; earmarking of land for (2) ship repairs at the small craft harbour and (5) for energy, liquid bulk, and gas storage; (4) widening of the IOT causeway; (6) the development of a new Berth 100 to increase handling capacity to 67 million tonnes per annum (tpa); (8) reconfiguration of Oil Jetty and Berth 104; (11) the development of Berth 205 (TNPA 2022) (Figure 4.6).

The medium-term port layout will include additional berth facilities in the Mossgas quay and Berth 200 to accommodate increased demand and changes in vessel; the infilling of land adjacent to the Oyster Dam (also called Seafarm Dam) to increase liquid bulk storage land area and the development of an liquefied natural gas (LNG) jetty for liquid bulk (TNPA 2022). Changes envisioned for the long-term layout (Figure 4.6) include (1) the development of additional berths or handling liquid bulk and LNG and other energy products; (2) Berth 199, 198 to accommodate increased demand and vessel size forecast (TNPA 2022).

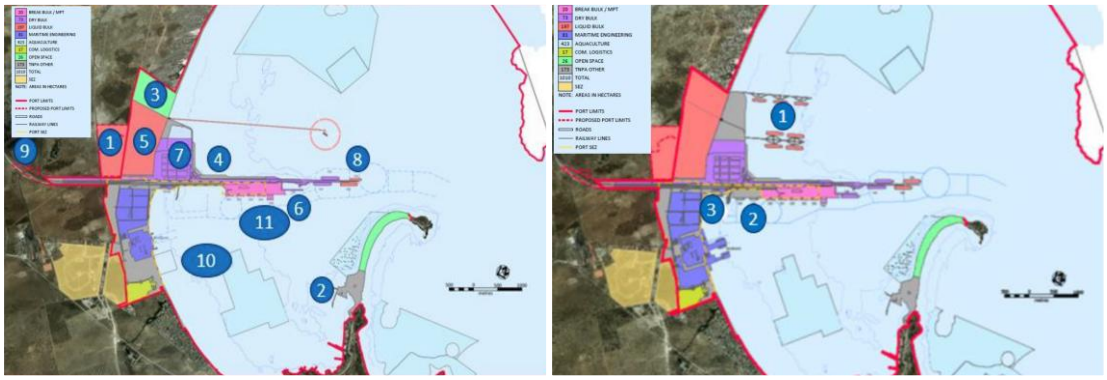

Figure 4.6. Layout of Transnet Saldanha Bay Port over (left) the short term and (right) the long-term (2032) (TNPA 2022).

Future industrial development of Saldanha Bay and the Port itself will be driven by Operation Phakisa, a government initiative launched in 2014 with the goal of boosting economic growth and creating employment opportunities under the National Development Plan 2030. Critical areas identified to further explore and unlock the potential of South Africa's oceans include marine transport and manufacturing, offshore oil and gas exploration, marine aquaculture and marine protection services and ocean governance. In line with this initiative, TNPA has 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.

4.3.2 Export of Mineral Ores

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 IOT (Figure 4.7), although smaller volumes have also been exported from the MPT (Figure 4.8). The Port 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 (see Section 4.6 below). Iron ore export volumes increased from 51.2 million tonnes in 2024/2025 to 55.4 million tonnes in 2025/2026 (Figure 4.7). The MPT has been primarily used for manganese exports since 2016 (Figure 4.8). South Africa accounts for the largest manganese reserves globally (approximately 78%), with most of the locally produced manganese exported for processing in China and production increasing substantially over time (Chamber of Mines South Africa 2017, Prager 2024). Manganese exports from the Saldanha Port increased in 2024/2025 to 6.35 million tonnes (Figure 4.8). Total metal exports from the MPT has increased over the last five years, driven by increased zinc exports, but has declined over the last two years from a peak in 2022/2023 (Figure 4.8).

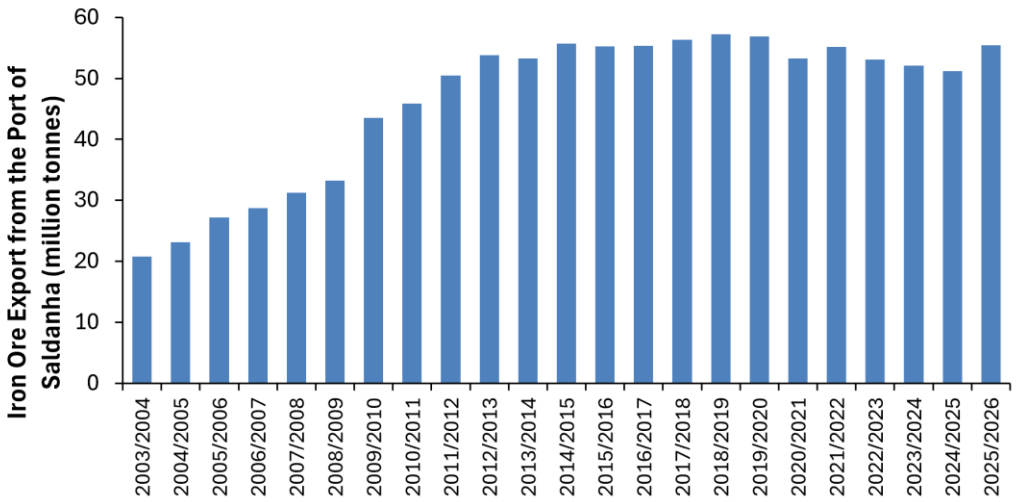

Figure 4.7. Annual exports of iron ore from the Port of Saldanha (Data provided by Transnet National Ports Authority 2025). Dates up to and including 2019/2020 represent Iron ore exported from the IOT. Dates from 2020/2021 onwards are total iron ore exports from both IOT and MPT. Data provided by Transnet National Ports Authority 2026

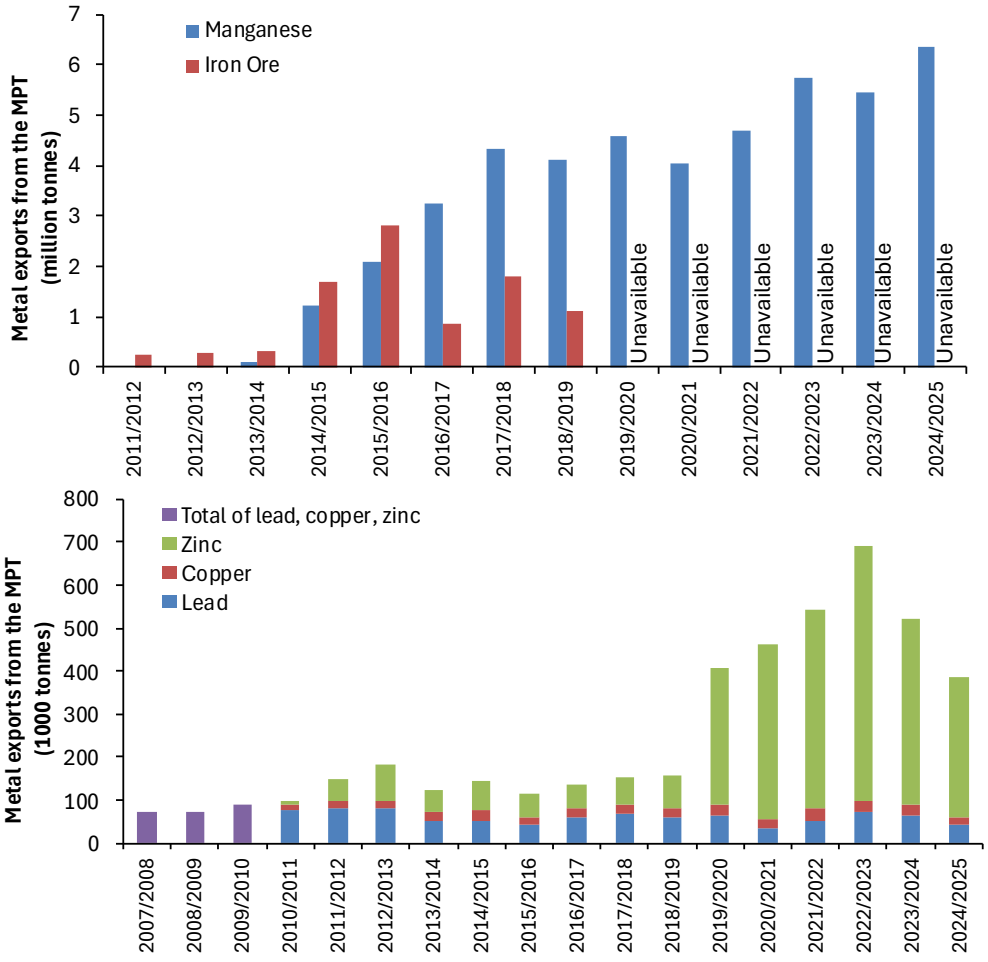

Figure 4.8. Annual exports of manganese and iron ore (top) and lead, copper, and zinc (bottom) from the Multi-Purpose Terminal at the Port of Saldanha Bay. Note that separate data for these commodities was only available for April 2010 to March 2024. Data provided by Transnet National Ports Authority 2026.

In 2022/2023, zinc exports peaked at ~595 200 tonnes but have since declined to 461 198 tonnes in 2025/2026 (Figure 4.8). The previous increase in zinc exports is linked to the opening of the Gamsberg opencast zinc mine by Vedanta Resources Limited. Lead exports have averaged approximately 56 436 tonnes a year over the last decade, with a drop to 26 473 tonnes in the 2025/2026 fiscal year (Figure 4.8). Copper is exported in small quantities compared to other metal ores and has also been in a decline over recent years, with exports in 2025/2026 totalling 11 840 tonnes, down from 17 944 tonnes the previous year (Figure 4.8). Rock phosphate exports have increased over recent years due to the opening of the Elandsfontein phosphate mine just outside Saldanha and now represents the fourth largest mineral export from the port. A record 363 472 tonnes of rock phosphate was exported in the 2025/2026 period, a 15% increase from the previous year's 315 698 tonnes (Figure 4.8).

4.3.3 Air Quality Management

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

The storage and handling of ore and coal not situated on the premises of a mine or works requires an atmospheric emission licence in terms of Chapter 5 of the National Environmental Management Air: Quality Act, 2004 (Act 39 of 2004) (as amended). TNPA submitted an application for a variation to the existing Atmospheric Emission Licence to increase the throughput from 60 to 67 million tonnes in June 2018, which included submission of an Air Quality Assessment Report (February 2018) and a public participation process. The application was denied by the Competent Authority on 12 September 2018, as the impact assessment report demonstrated that during the monitoring period, National Dust Control Regulations for residential and non-residential fallout dust rates (600 and 1 200 mg/m^2 per day, respectively) were exceeded. It was therefore concluded that cumulative impacts going forward would be unacceptable considering the current impact of dust emissions.

A Provisional Atmospheric Emission Licence for the storage and handling of manganese at the MPT was issued by the DEA on 26 September 2018 (Reference: AEL/WCP/TPT/26/06/2018-2387). The Provisional Atmospheric Emission Licence was appealed in November 2018, the main concern being the "*harmful and health effects of manganese to people, water, aqua farms, tourism and businesses including the efforts to develop Saldanha Bay as a Green City and that an EIA should have been conducted*". The appeal was upheld (January 2020, Reference LSA 177442) and the Provisional Atmospheric Emission Licence for the storage of manganese ore was set aside. The upheld appeal decision also identified that the activities required for the expansion of current manganese in the port would have triggered an EIA prior to the issuing of the Provisional Atmospheric Emission Licence. Given this, the quantity of manganese being stored at the MPT has been significantly reduced.

There are also a number of small operations in the SBM that do not require an Atmospheric Emission Licence (i.e. storing and handling ore of less than 100 000 tonnes)

that have resulted in significant cumulative impacts on air quality, including but not limited to the uncovered transportation of materials through residential areas by rail or road.

The cumulative impacts on air quality over the years have continued to upset residents and businesses and multiple lobby groups have formed both formally and informally. A prominent current example is the Red Dust Action Group (RDAG), formed in 2015 as a community driven Facebook lobby group to pressure authorities on air quality and emissions related to the ore exports. In 2024 the RDAG was registered as a non-profit organisation and in 2025 launched an official website where concerned citizens could register as members and donate to the organisation. Recently, the RDAG working with the University of Cape Town launched an AI-driven WhatsApp reporting tool to make it easier for residents to file complaints and upload evidence of impacts of red dust in the area. At time of writing the group's website (https://rdag.org.za/) stated having 3252 members. Information on numbers of legitimate complaints logged was requested from the group but no response was received by time of publication.

The WCDM gave notice on 15 June 2026 of rejecting Transnet's application for renewal of an Existing Atmospheric Emission Licence (WCWD013) citing persistent red dust pollution and community complaints. However, the WCDM withdrew their refusal weeks later stating that the matter should rather be handled by the national government instead of the district level, prompting a major shift in the review process. The refusal withdrawal by WCDM did not imply acceptance or approval of said license. The new competent licensing authority will be the Department of Forestry Fisheries and the Environment (DFFE).

4.3.4 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 & Robin Lewis 2006). Dredged material may enter the water column 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, Erftemeijer & Robin Lewis 2006). 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 sediment. Dredging in Saldanha Bay has had tremendous immediate impact on benthic micro- and macrofauna, through both direct disturbance and as a result of particles suspended in the water column (especially affecting suspension feeders like fish and zooplankton). These suspended particles can also limit the penetration of sunlight in the water column, causing die offs of algae and phytoplankton. Furthermore, resuspended fine sediment could potentially 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. Dredging can also result in the resuspension of toxic metals, hydrocarbons and organic matter trapped in the sediment, with direct toxicity impacts through bioaccumulation as well as implications for oxygen levels. 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 South Africa's largest and deepest natural port and as a result has undergone extensive harbour development and several bouts of dredging and marine blasting (summarised in Table 4.1). In June 2025, the Port completed dredging operations at the Mossgas quay, with a total volume of approximately 20 000 m^3 of material removed (Dimakatso Mothibeli *pers. comm.* 2025). No further dredging activities have taken place since then.

Table 4.1. Summary of dredging activities in the Port of Saldanha (1974–2025).

Date	Details
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 removed for the expansion of the MPT.
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 removed for maintenance of the Mossgas quay and MPT.
2009–2010	7 300 m^3 of sediment removed to allow for the establishment of a new ore-loading berth. Maintenance dredging (unknown quantity) conducted by the South African National Defence Force (SANDF) at the Salamander Bay boatyard.
2015–2016	25 000 m^3 Expansion of the General Maintenance Quay.
2019	14 265 m^3 of sediment removed for maintenance of the Off-Shore Supply Base quay.
2019–2020	6 403 m^3 of sediment removed for maintenance of the Mossgas channel.
2020	13 433 m^3 of sediment removed for maintenance of the Off-Shore Supply Base Berth channel.
June to September 2022	3 228 m^3 removed for maintenance of the Off-Shore Supply Base Berth and 15 170.67 m^3 for maintenance of the Mossgas Berth.
November 2022 to March 2023	6 491.09 m^3 sediment removed for maintenance of the Off-Shore Supply Base Berth.
June 2025	20 000 m^3 of material removed at the Mossgas quay.

4.4 SHIPPING ACTIVITIES

4.4.1 SHIPPING VOLUME

Since 1994/1995, the total number of vessels entering the Port of Saldanha has continued to increase steadily over time (despite some fluctuations between years), averaging ~560 vessels/year over the last decade (Figure 4.9), and ~47 vessels per month (Figure 4.10). Fewer total vessels (538 ships) were recorded in 2025/2026 (average of ~45 vessels per month), the majority of which remains dry bulk vessels (Figure 4.9) associated with the export of minerals like iron ore, manganese, zinc and rock phosphate. This is despite the overall increase of almost 7% in exports from the Saldanha port over the 2025/26 period, and the highest ever Manganese exports from the MPT (see Figure 4.7 and Figure 4.8). It isn't clear how these changes correlate and additional information was sought from Transnet, but no reliable explanation could be sourced by time of publication.

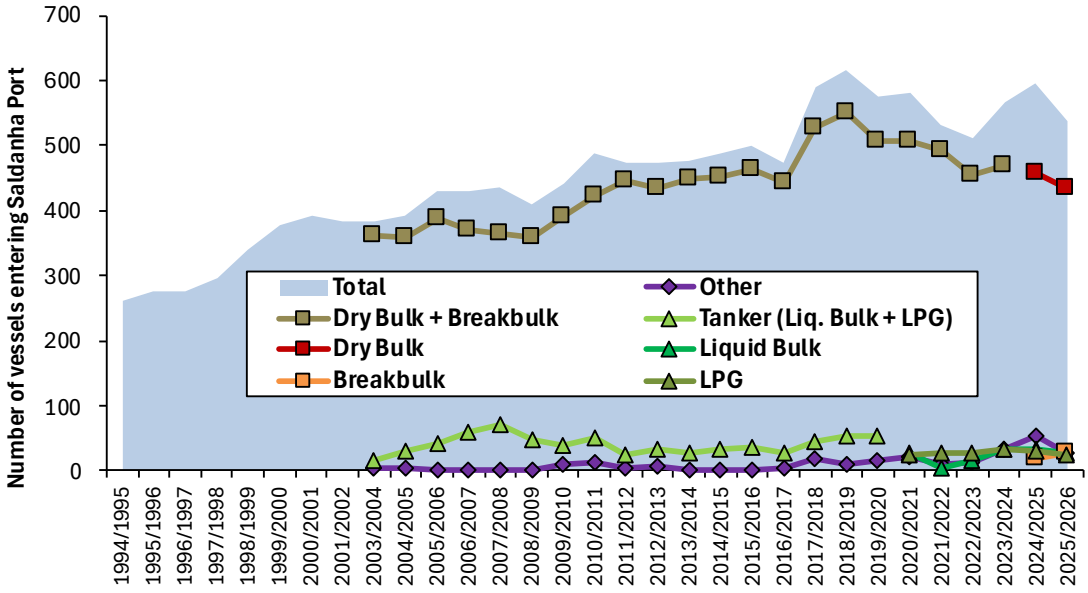

Figure 4.9. The numbers and types of vessels entering Saldanha Port per year. About 10% fewer total number of vessels were recorded in the 2025/2026 period despite an increase in export volume. The total number of vessels entering the Port is shown as the blue area. Before the 2003/2004 period only the total number of vessels were recorded. All liquid carrying vessels have triangular markers and in shades of green. The number of Liquid bulk and LPG vessels are reported separately from 2024/2025 period being previously only recorded as "Tanker" vessels. Similarly, Dry Bulk and Breakbulk vessels are reported separately from 2024/2025 period being lumped together before. All these have square markings. Uncategorised "Other" vessels have diamond shaped markings (Sources: Marangoni (1998), Awad et al.(2004), and TNPA unpublished data 2003–2026).

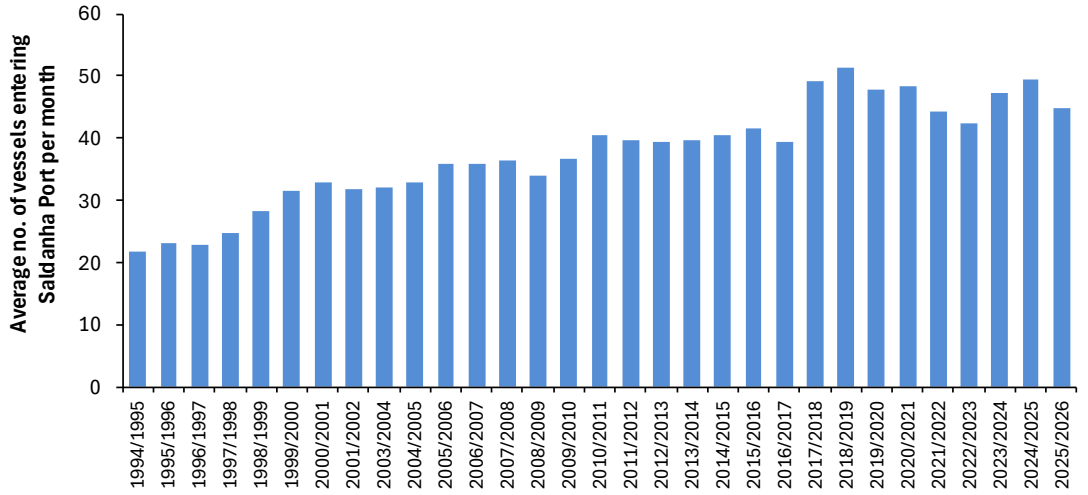

Figure 4.10. The average number of vessels entering Saldanha Port per month. TNPA unpublished data 2003–2026.

Increases in shipping traffic come with a number of associated risks, especially in a port environment, where the risks of collisions and breakdowns increase owing to increased vessel traffic, difficult vessel manoeuvres, and the discharge or uptake of ballast water in lieu of cargo that has been loaded or unloaded.

4.4.2 BALLAST WATER DISCHARGES

Discharge of ballast by vessels visiting the IOT poses a risk to the health of Saldanha Bay and Langebaan Lagoon. Ballast water is a primary vector for the introduction of alien and invasive marine species (Robinson et al. 2020). Ballast water is used to maintain trim and stability in ships, especially when partially loaded or empty. It is pumped on board into the holds or ballast tanks of a ship in whatever port cargo is unloaded. Ballast water usually contains a certain amount of sediment as well as many species of living marine organism from the port of origin (Figure 4.11).

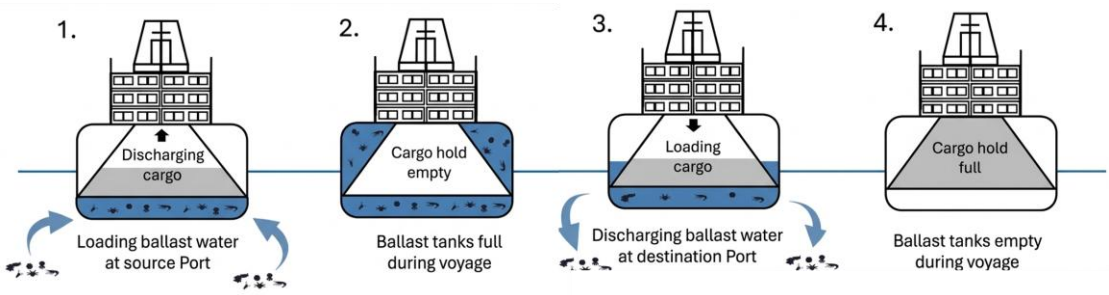

Figure 4.11. Cross section of a ship showing ballast tanks and ballast water cycle. Adapted from GloBallast, International Maritime Organization.

Concentrations of trace metals (cadmium, copper, zinc, and lead, etc.) in excess of South African Water Quality Guideline (WQG) limits have been detected in ballast water and ballast tank sediments from ships deballasting in Saldanha Bay (Carter 1996). These discharges are almost certainly contributing to trace metal loading in the water column and are indicated by their concentration in filter-feeding organisms in the Bay (refer to Chapter 6: Water Quality).

Ballast water discharges can, however, be effectively managed. To address environmental impacts and risks, the International Convention for the Control and Management of Ship's Ballast Water and Sediments of 2004 (Ballast Water Management Convention). The programme entered into force in 2017 (IMO 2017), and has 63 Contracting Parties, representing 68.5% of world merchant shipping tonnage. The programme presents a number of strategies to minimise risk, including minimising uptake of organisms into ballast water tanks (i.e. avoiding ballast water uptake in shallow and turbid areas, and avoiding uptake at night when many organisms migrate vertically to feed) and ballast water exchange (this involves the exchange of ballast water between ports, in mid-ocean and in deep water, in order to reduce the risk of organisms carried in the water finding a suitable environment on discharge). While South Africa ratified this Convention and published the Draft Ballast Water Management Bill in 2013 (Government Gazette Notice 340 of 2013), this Draft Bill has not been gazetted. While no domestic legislation is currently in place to regulate ballast water discharge, the TNPA in Saldanha Bay has implemented a number of mechanisms to track and control the release of ballast water into the harbour. For example, there is a two-phase ballast water release protocol for ships visiting the Port of Saldanha—the first release occurs upon entering Saldanha Bay (i.e. Big Bay), and the second once the ship is berthed and loading (Awad et al. 2004). As a result, as much as 50% of the ballast water is released in the vicinity of the IOT on 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 tripled between 1994 and its peak in 2018/19 (261 to 616 vessels per annum) (Figure 4.9)[2]. This, along with increases in average vessel size, more than tripled the volume of ballast water discharged annually between 1994/5 and 2020/2021 (from 8.2 to 25.9 million m^3) (Figure 4.12). In 2024/2025 however, there was a decline in the discharge ballast volume (down to 17 million m^3), some 30% below average of 23.2 million m^3 for the decade prior to 2020 (Figure 4.12), despite the overall increase in vessel traffic (Figure 4.9 and Figure 4.10).

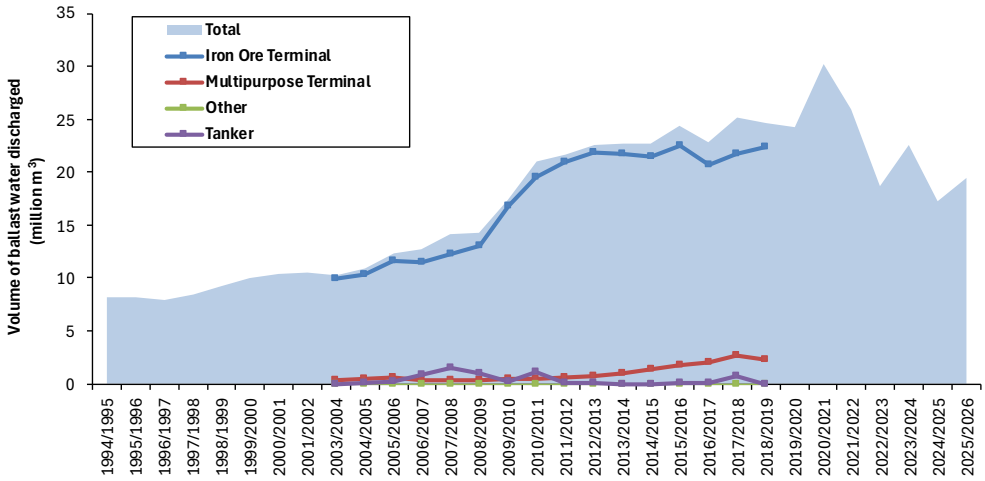

Figure 4.12. Volumes of ballast water discharged into Saldanha Port per rolling year. Volume = blue area. Ballast water discharge per vessel type was reported between 2003/2004 and 2018/2019. Iron ore terminal here likely represents dry bulk vessels while Multipurpose terminal vessels represent Breakbulk vessels. Tankers are exclusively liquid bulk vessels with some LPG vessels between 2017–2019 (Sources: (Marangoni 1998, Awad et al. 2004), TNPA unpublished data 2003–2026).[3]

Average ballast water discharge volumes based on vessel types were only available between 2003/2004 and 2018/2019. Vessels docking at the IOT had a higher average volume of ballast water discharge than other vessel types, with volumes increasing from 54 400 m^3 per vessel in 2003/4 to 71 200 m^3 in 2018/19 (Figure 4.13). Generally, iron ore tankers contributed approximately 50% to the observed vessel traffic (Figure 4.9) and 91% to the total water discharged (Figure 4.12). This is to be expected given that iron ore tankers are large vessels and hold the highest quantities of ballast water. While discharge volumes for tankers fluctuates irregularly between 0 m^3 and 21 800 m^3, vessels docking at the MPT increased from less than 3 000 m^3 in 2010/11 to 9 600 m^3 in 2018/19 (Figure 4.13).

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2 In the 2025 report it was speculated that the total volume of ballast water data provided by TNPA in the 2020/21 was likely erroneous due to it dropping to 9.8 million m^3 (40% of the previous year's total volume (Figure 4.12). Investigation since found formatting errors data received and have been corrected.

3 Data received from TNPA have been reported in various different manners over the years and some reporting periods or classifications contained some flaws that rendered detailed analyses, and consequently interpretations difficult. For example, for an extended period of time TNPA reported all ships visiting the MPT as "Breakbulk" ships and all visiting the IOT "Dry bulk" or "Bulkers". At a glance this makes sense as only Dry bulk carriers ever visit the IOT and the MPT is the only terminal that can service Breakbulk vessels. However, the MPT can be used for more than servicing breakbulk vessels. Indeed, it has always been the case that a substantial number of vessels docking at the MPT are dry bulk vessels. Therefore,the detailed vessel breakdown may be over- or underreporting in some areas.

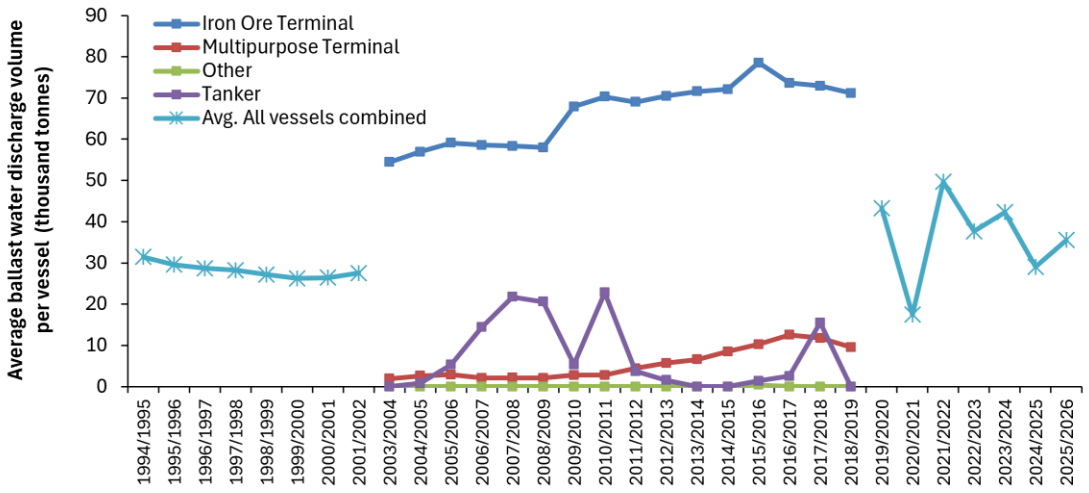


Figure 4.13. Average ballast water volumes discharged per vessel into Saldanha Port per year. The average total volume of ballast water discharged in Saldanha Port is shown as the light blue line. Data for the different types of vessels are only available from 2003 to 2019. Sources: (Marangoni 1998, Awad et al. 2004), TNPA unpublished data (2003–2026).

4.4.3 SHIPPING RELATED NOISE

Underwater noise from ships is a widespread form of anthropogenic disturbance that can mask important biological sounds, alter behaviour, and stress marine life. Effects range from short-term disruption of communication to potential long-term impacts on fitness and population dynamics and chronic noise exposure can elevate stress indicators (e.g. cortisol-like responses in some species) and may affect immunity, growth, and reproduction. Marine mammals like whales, dolphins, pinnipeds are highly affected due to reliance on acoustic communication and echolocation. Masking, hearing damage, and behavioural disruption are primary concerns. Furthermore, some fish and invertebrate species exhibit altered schooling, spawning behaviour, or habitat preference and nocturnal and coastal species with delicate acoustic cues can be particularly vulnerable.

International bodies (e.g. IMO) and various NGOs emphasise reducing shipping noise as feasible and beneficial for ecosystem health. Mitigation approaches include quieter propeller designs, speed reductions in sensitive areas, route planning to avoid critical habitats, real-time noise monitoring, and operational measures like hull maintenance to reduce cavitation noise.

Anthropogenic ocean noise generated by commercial shipping consists predominantly of continuous, low-frequency sound emissions (20–500 Hz) that propagate across extensive marine areas. Because this frequency spectrum directly overlaps with the primary acoustic communication and sensing channels of many marine vertebrates, vessel traffic induces widespread acoustic masking, behavioural disruption, and habitat displacement. In a foundational study on marine mammals, Rolland et al. (2012) provided the first empirical evidence linking vessel noise directly to chronic physiological stress. By analysing faecal glucocorticoid metabolites in North Atlantic right whales following a sudden reduction in commercial ship traffic, the researchers demonstrated that a 6 dB drop in low-frequency vessel sound resulted in a significant decrease in baseline stress hormone levels, confirming that routine shipping traffic induces persistent, systemic physiological stress in large marine vertebrates.

Beyond marine vertebrates, anthropogenic underwater noise severely impacts marine invertebrates, which lack specialized acoustic organs but detect particle motion and physical vibrations via statocysts and chordotonal sensilla. Exposure to low-frequency vessel noise causes physiological disruption, energetic strain, and impaired cognitive decision-making during critical life functions. In a high-impact experimental study on decapod crustaceans, Wale et al. (2013) demonstrated that playbacks of commercial ship noise significantly disrupted feeding behaviour and compromised anti-predator responses in shore crabs *Carcinus maenas*. Noise-exposed individuals were significantly slower to seek shelter following simulated predator attacks and displayed disrupted foraging efficiency, demonstrating that continuous shipping noise imposes direct fitness costs through heightened predation vulnerability and elevated metabolic energetic demands.

Applying these findings to Saldanha Bay on South Africa's West Coast highlights a significant environmental concern. As a semi-enclosed, highly industrialized deep-water port hosting heavy commercial iron ore carriers, oil tankers, and marine industrial infrastructure, Saldanha Bay forms an acoustic environment where low-frequency shipping sound reverberates within shallow estuarine waters. Assessing regional shipping noise impacts, Pichegru et al. (2022a) revealed that doubling maritime traffic and vessel bunkering in South African coastal waters doubled ambient sound intensity, which directly correlated with an 85% collapse of local African penguin *Spheniscus demersus* breeding populations due to acoustic masking and severe disruption of collective foraging behaviour.

While site-specific bioacoustic research in Saldanha Bay remains scarce, several resident and regularly occurring species can be flagged as plausible candidates for elevated risk, based on their sensory biology, life-history requirements, and spatial overlap with the Bay's shipping lanes and anchorages. The Cape fur seal *Arctocephalus pusillus pusillus*, which maintains a permanent breeding colony on islands within the Bay itself, is a strong candidate: pinnipeds have good underwater hearing sensitivity across the low-frequency range dominated by vessel traffic, and chronic exposure could mask the vocal contact calls that mother-pup pairs rely on for reunion and provisioning, in a colony situated only a few kilometres from the main shipping channel. The African penguin, already implicated above by regional evidence linking vessel noise to population collapse (Pichegru et al. 2022a), remains similarly exposed given its non-migratory, colonially breeding island populations within the Bay itself.

Among cetaceans, the endemic Heaviside's dolphin *Cephalorhynchus heavisidii*, which frequents the shallow coastal waters of the Benguela ecosystem including Saldanha Bay, presents a more nuanced case. Its echolocation clicks are narrow-band, high-frequency signals centred near 125 kHz with little energy below 100 kHz (Morisaka et al. 2011), largely outside the 20–500 Hz band dominated by shipping noise, which may reduce direct masking of its foraging sonar. However, the species also produces lower-frequency, broadband communication clicks used to maintain contact within social groups, and these fall within a range more susceptible to masking by vessel noise, potentially compressing an already short active communication space.

Among invertebrates, the West Coast rock lobster *Jasus lalandii* warrants particular attention. Saldanha Bay is a documented site of substantial puerulus (postlarval) settlement and early juvenile recruitment (Groeneveld et al. 2010), and, like other decapods, settling pueruli likely rely on substrate-borne vibration and other physical

cues to select suitable habitat. Given that the closely related Mediterranean spiny lobster *Palinurus elephas* shows measurable behavioural and physiological stress responses (including elevated locomotor activity, glucose, and Hsp70 expression) following exposure to boat noise (Filiciotto et al. 2014), a comparable sensitivity in *J. lalandii*, already a fishery under long-term exploitation pressure, cannot be discounted, particularly during this vulnerable settlement window. Soniferous fish species such as the sciaenids frequenting South Africa's West Coast (e.g. silver kob *Argyrosomus inodorus*) may also be disproportionately affected as their swim-bladder drumming calls, used to coordinate spawning aggregations, typically fall within the 100–500 Hz range, overlapping almost entirely with the dominant shipping noise band (Slabbekoorn et al. 2010), raising the possibility that chronic vessel noise could interfere with mate attraction and spawning success in an already heavily exploited species. Finally, the extensive mussel and oyster mariculture operations within the harbour precinct place farmed bivalves in close, continuous proximity to vessel traffic; although the effects of chronic vibration exposure on bivalve physiology remain poorly studied generally, this represents both an ecological and economic dimension worth investigating locally.

Given the near-total absence of empirical bioacoustic data for Saldanha Bay, these associations remain speculative extrapolations from studies conducted elsewhere, informed by each species' known sensory ecology and its spatial and temporal overlap with vessel activity in the Bay. They are intended to flag priority candidates for future investigation rather than to assert confirmed impacts and underscore the need for baseline underwater soundscape monitoring and species-specific studies within Saldanha Bay itself.

4.4.4 Oil spills

An increase in the incidence and risk of oil spills is also associated with this increase in shipping traffic. In response to this risk, the “Port of Saldanha oil spill contingency plan” was published in 2007 by TNPA and Oil Pollution Control South Africa. The plan is designed to interface with the “National oil spill contingency plan” and with the “Terminal oil spill contingency plan” to ensure a rapid response to oil spills within the port itself and by approaching vessels with a three-tiered response (Table 4.2) (TNPA 2007). 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 (Martin Slabber (SAMSA) *pers. comm.*). Specialized facilities and equipment for marine oil spill containment, shore decontamination, and oiled seabird rehabilitation are coordinated under the South African Maritime Safety Authority (SAMSA) and the Department of Forestry, Fisheries and the Environment (DFFE) through national and local contingency plans. Nevertheless, given the high environmental sensitivity and restricted water exchange within Langebaan Lagoon, risk mitigation and spill prevention remain the central objective for port and coastal management (Kock et al. 2025) (SAMSA 2020; SANParks 2024).

While no major oil spills have occurred in Saldanha Bay to date (Martin Slabber (SAMSA) *pers. comm.*), minor spills do occur, which have the potential to severely impact the surrounding environment. The most recent minor incident happened in October 2023. The Paraclete vessel partially sank (but was still buoyant) near the Government Jetty at

the Saldanha Bay fishing harbour, carrying around 300 l of hydraulic oil and an unknown amount of engine oil. All debris (and associated potential for oil spill) was removed and there was no major pollution incident reported at Pepper Bay resulting from this vessel sinking or salvage.

Table 4.2. Oil spill tiers and responses as per the Port of Saldanha oil spill contingency plan (TNPA 2007).

Tier	Definition	Response
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.
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.
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.

4.4.5 Other shipping impacts (Ship strikes, biofouling, invasives)

Shipping activity in Saldanha Bay introduces further physical and biological pressures on marine and coastal fauna: direct vessel collisions, and the transport and establishment of fouling and invasive organisms on vessel hulls and in ballast water.

Ship strikes, collisions between vessels and large whales, are increasingly recognised as a significant, if poorly quantified, source of mortality worldwide. Laist et al. (2001) established that lethal or severe injuries are disproportionately caused by vessels 80 m or longer travelling faster than 14 knots, with whales frequently unable to detect or avoid an approaching hull in time. Along the South African coast, ship strikes and entanglements have been documented as a persistent, if historically low-incidence, source of mortality in southern right whales *Eubalaena australis* over multiple decades (Best et al. 2001, Vermeulen et al. 2022), though both studies caution that the true toll is likely underestimated: struck whales frequently sink offshore rather than stranding, leaving no carcass to record. This concern has sharpened considerably in the past two years. A preliminary risk assessment presented to the International Whaling Commission in 2026 by researchers from the University of Pretoria's Whale Unit, Afriseas Solutions and the International Fund for Animal Welfare found extensive spatial overlap between shipping corridors and the habitats of six baleen whale species along South Africa's southwest coast, explicitly identifying the approaches to Cape Town, Saldanha Bay and Cape Agulhas as zones of elevated strike risk (Vermeulen et al. 2026, cited in Bega (2026)). The number of large, fast-moving vessels (>15 knots) transiting these waters has quadrupled since late 2023, as international shipping has rerouted around the Cape of Good Hope in response to instability in the Red Sea and the Strait of Hormuz. Saldanha Bay's status as a major bulk-carrier terminal places it directly within this corridor, and the assessment further noted that port-approach zones, where vessels necessarily converge and cannot be rerouted offshore, pose a particular risk to feeding whales, including the large humpback whale "supergroups" that periodically aggregate off the West Coast and

may be less attentive to approaching vessels while foraging. Southern right, Bryde's *Balaenoptera edeni* and humpback whales showed the greatest modelled risk reduction (31–50%) from even modest (<20 nautical mile) offshore shifts in shipping lanes, suggesting that route and speed management in the vicinity of Saldanha Bay could meaningfully reduce strike risk without materially lengthening voyages.

Biofouling is another shipping related impact which has garnered more attention in recent decades. Vessel hulls, sea chests and other submerged structures provide substrate for the attachment and transport of fouling organisms like algae, barnacles, mussels, ascidians, bryozoans and associated mobile fauna between ports. As one of South Africa's largest bulk ports, combined with intensive mussel and oyster mariculture operations within the harbour itself, Saldanha Bay presents an elevated risk of fouling-organism introduction and establishment (Awad et al. 2003, Havenga 2014). The mechanisms, monitoring approaches and ecological consequences of biofouling in the Bay are addressed in detail in Chapter 11 and are therefore only flagged here as a further vector by which shipping activity affects the local marine environment.

The introduction of invasive species through shipping activities is a well-studied field. Biofouling and ballast water discharge are the primary vectors by which Saldanha Bay has become a documented point of entry for marine alien species in South Africa. The best-known case is the Mediterranean mussel *Mytilus galloprovincialis*, first recorded in the Bay in the late 1980s and since spread along the west and south coasts, where it has displaced indigenous species and altered intertidal community structure (Robinson et al. 2004a). This is not a historical, one-off event: continued surveillance has since recorded further shipping-associated introductions in the Bay, including two South American invertebrates detected on Saldanha's piers in 2015 and 2017 (Peters & Robinson 2018a), underscoring that the Bay remains an active introduction hotspot under current shipping volumes. As with biofouling, invasive species are covered in detail in Chapter 11.

4.5 The Saldanha Bay Industrial Development Zone

4.5.1 Introduction

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 IDZ in October 2013. The Saldanha Bay IDZ is the first Special Economic Zone (SEZ) to be located within a port and is the only sector specific Special Economic Zone in South Africa catering specifically to the oil and gas, maritime fabrication and repair industries and related support services (SLR 2019). The Saldanha Bay IDZ is managed by the Saldanha Bay IDZ Licensing Company (LiCo). The Saldanha Bay IDZ LiCo is the holder of an Environmental Authorisation (EA) for the development of an oil and gas offshore service complex (granted 16 November 2015).

The Saldanha Bay IDZ 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, see Section 4.4 above), increased pollution of the Saldanha Bay through maintenance and repair activities, and stormwater runoff.

4.5.2 ENERGY DEVELOPMENTS

GREEN HYDROGEN HUB

"Green hydrogen" is a clean energy source that has the potential to play a crucial role in decarbonizing energy sectors by replacing the use of fossil fuels. Hydrogen gas (H_2) is generated via the process of electrolysis and involves splitting water molecules into H_2 (or simply referred to as hydrogen) and oxygen (O_2). Unlike traditional methods of hydrogen production that rely on fossil fuels, green hydrogen is produced using electricity from renewable sources (such as solar or wind power) and does not emit greenhouse gases during its production. However, as the production of hydrogen requires freshwater, Reverse Osmosis (RO) seawater desalination has been proposed as a solution for green hydrogen projects in the country, and any Green Hydrogen plant in South Africa will also require the construction of a desalination plant.

In October 2022 the Saldanha Bay IDZ signed a memorandum of understanding with Sasol South Africa to facilitate a green hydrogen hub in Saldanha Bay and research is underway to investigate the economic, technical, legal, environmental, and social suitability of desalination plants as a water source for green hydrogen projects in South Africa. Any proposed development of a renewable energy desalination plant project that triggers listed activities in accordance with the amended EIA regulations, of the National Environmental Management Act (NEMA) 107 of 1998 will require the completion of a Scoping and EIA.

Solar Capital, a subsidiary of Phelan Green Energy, have initiated a Saldanha green hydrogen project. In October 2023, the project which included the construction of a green hydrogen and ammonia complex was granted Strategic Integrated Project status (Barradas 2024). The project is expected to generate 2.5 GW using a combination of wind and solar power, for the production of 85 000 t/y of green hydrogen for use in industrial applications (Barradas 2024, Claasen 2024). The project was expected to start green hydrogen production and export in early 2026, with approximately 2 500 jobs created in the construction phase and 500 permanent jobs during operation (Barradas 2024, Claasen 2024).

Phelan Green signed a supply agreement with Sungrow Hydrogen in mid-2026 for the first 200 MW phase of a planned 800 MW electrolyser deployment to produce hydrogen for creating electro-sustainable aviation fuel. Construction of this phase is planned to start in the fourth quarter of 2026. The planned operational output of the final facility is 140 000 tonnes of electro-sustainable aviation fuel per year.

FRONTIER POWER SA GAS TO POWER FACILITY

The development of the Gas to Power facility proposed by Frontier Power SA reporting in previous State of the Bay reports has stalled or been shelved, mirroring the wider suspension and cancellation of various port-based LPG Floating Storage Regasification Unit (FSRU) and gas-to-power pipelines in Saldanha Bay.

DEVELOPMENT OF LIQUID PETROLEUM GAS FACILITIES IN SALDANHA BAY

Avedia Energy (Pty) Ltd developed a land based liquid petroleum gas storage facility on Portion 13 of Farm Yzervarkensrug No. 127 in Saldanha. The storage facility was designed to include 16 mounded bullet tanks with a storage capacity of 250 tonnes each (Frans

Lesch, ILF Consulting Engineers, Project Manager at Avedia Energy Saldanha LPG plant *pers. comm.* 2015) (Clark et al. 2014). Avedia Energy completed construction of their LPG storage facility in 2017 and upgraded the facilities to support an additional 6 000 tonnes of LPG monthly in the second half of 2018. In 2022, four liquefied petroleum gas (LPG) storage tanks were delivered to Avedia in 2022, which increased the storage capacity by an additional 8 000 tonnes.

The construction of the LPG import facility in the Saldanha Bay Harbour, by Sunrise Energy (Pty) Ltd, was completed in 2016. This development aimed to supplement existing 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. 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 WQT monitoring program, and sediment sampling adjacent to the marine site has been conducted annually for several years now. In December 2022, shareholders in Sunrise Energy, Ilitha Group Holdings Proprietary applied to put the company into business rescue, detailing that the Sunrise Energy business has been making a loss from 2017, when it was first commissioned.

REFINED FUEL STORAGE TERMINAL AND ASSOCIATED INFRASTRUCTURE

Cape Ocean Terminals (COT) was granted Environmental Authorisation (16/3/3/2/F4/17/3004/19) in 2019 for the development of a refined fuel storage terminal and associated infrastructure on a portion of portion 23 of the farm No. 127 in the Saldanha Bay IDZ (called 'Project Albatross'). This EA was amended in 2021 (16/3/3/5/F4/17/3004/21-B) and again in 2024 (16/3/3/5/F4/17/3018/24) to allow for a total fuel storage capacity of 250 000 m^3 at Cape Ocean Terminals' planned bulk refined fuel storage facility at the Saldanha Bay IDZ.

GAS FIRED INDEPENDENT POWER PLANT

The International Power Consortium South Africa (Pty) Ltd have proposed the construction of a Combined Cycle Gas Turbine power plant (1 507 megawatt (MW) net capacity) as a LNG power supply for Saldanha Steel (now part of ArcelorMittal) (ERM 2016). LNG would be supplied by ship to the Port of Saldanha, where it would 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 EA from the then National Department of Environmental Affairs (DEA) under the NEMA through a Scoping and EIA process on 24 February 2017 but the plans for the LNG power plant were ultimately scrapped in 2025.

CRUDE OIL TERMINAL AND STORAGE FACILITY

Oiltanking MOGS Saldanha (RF) (Pty) Ltd, a joint venture between Oiltanking GmbH, MOGS (Pty) Ltd and Industrial Development Corporation of South Africa Ltd announced successful commissioning of a new crude oil terminal with the successful receipt of approximately 900 000 barrels of crude oil on 17 April 2020 (Oiltanking.com 2020).

Environmental authorisation was first granted by the DFFE in 2019 (16/3/3/2/F4/17/3032/18). The first phase of the terminal comprising nine tanks with a total storage capacity of 9.9 million barrels has been operational since 2020 with a further three tanks under construction.

4.6 THE SISHEN-SALDANHA ORE LINE EXPANSION PROJECT

Iron ore mined in Hotazel, Postmasburg and Sishen is 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 installed a third iron ore tippler to facilitate iron ore exports of 60 million tonnes per annum (tpa) (GIBB 2013, Venter 2019).

While the line is currently limited to transporting 60 million tonnes of iron ore as per the current air emission licence, negotiations are ongoing to obtain a new licence to increase this to 67 million tonnes of iron ore (Transnet Freight Rail 2023). In addition, the Sishen-Saldanha ore line expansion project proposes the expansion of the ore line to increase the capacity of the current Sishen-Saldanha railway and port to 88 million tpa to satisfy the global demand for iron ore.

Transnet recently increased the length of the manganese ore train from 312 to 375 carriages, thereby increasing the volume of manganese transported during individual trips from roughly 19 500 tonnes to roughly 23 500 tonnes. An increase in rail capacity will result in a greater volume of ore arriving in Saldanha and accordingly an increase in ship traffic will be necessary to transport this product globally (see Section 4.8).

4.7 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 dominant application of phosphorus is in fertilisers, with growing demand in the agricultural sector (Braaf 2014)

The proposed mining area is located on the Elandsfontein Aquifer System (EAS) and near the Langebaan Road Aquifer System. These aquifer systems are defined by palaeo-channels that have been filled with gravels of the Elandsfontein 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) (Julian Conrad (GEOSS) *comm*. 2016). The EA was granted in February 2015 and a Water Use Licence (WUL) in April 2017.

In December 2021, the Elandsfontein phosphate processing plant, operated by Kropz Elandsfontein, received its first input of ore aimed at initiating the processes required for the production of the first concentrate. Commissioning activities were expected to transition into full scale ramp-up of the mining and beneficiation plant before the end of 2022, but there has been a delay due to “ore variability in the current mining area” (Kropz Plc 2024).

Transnet provided Kropz Elandsfontein with a draft port access agreement to support the long-term export of the company’s phosphate rock through the port of Saldanha in 2021,

with the agreement also allowing for the exporting up to 30% of the output from the Port of Cape Town, in case the capacity through Saldanha is not available (NS Energy 2021). While the first phosphate rock ore exports were expected to commence in the March 2022, export data from recent fiscal years did not report any phosphate exports (see Section 4.3.2). However, most recent data show rock phosphate as the current fourth largest export from the Port of Saldanha with Kropz still increasing both its rock phosphate grading and output.

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

4.8 MANGANESE STORAGE EXPANSION

A maximum of 90 000 tonnes of manganese is currently being stored in two sheds at the MPT, with current annual throughput at around 400 000 tpa. TNPA applied for EA to expand its operations and increase the storage of manganese in dedicated storage areas at the terminal to 450 000 tonnes, increasing the annual throughput to 8 million tonnes. This expansion is intended to supply in the global demand of manganese and in doing so provide major socio-economic benefits to the Western Cape and South Africa. The DFFE granted the EA to increase the manganese storage and handling capacity at the MPT on 13 October 2022. A total of 21 appeals were received by the DFFE in November 2022, but in the appeals decision letter (LSA 223883) dated 3 July 2023 Minister Creecy published her decision to dismiss the appeals and uphold the approval of the EA.

4.9 TNPA PROJECTS UNDER 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 TNPA proposed the development of new infrastructure at the Port of Saldanha in line with the objectives of Operation Phakisa, namely a Vessel Repair Facility for ships and oil rigs (Berth 205), a 500 m-long jetty at the Mossgas quay, and a floating dry dock for inspection of Offshore Supply Vessels.

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 volume 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 to Section 4.3.4).

4.10 EFFLUENT DISCHARGES INTO THE BAY

4.10.1 OVERVIEW

Disposal of wastewater is a major problem in the region, and much of it finds its way into the Bay as partially treated sewage, stormwater, industrial effluent (brine, cooling water discharges and fish factory effluent) and ballast water. To date, six Coastal Waters Discharge Permits (CWDPs) have been issued to companies discharging effluent into Saldanha Bay with three of these having lapsed/the operation has closed down (Table 4.3). Six General Discharge Authorisations (GDAs) have also been issued (Table 4.3).

Table 4.3. Applications for Coastal Waters Discharge Permits and issued permits for effluent discharges into Saldanha Bay (Source: DFFE 2026).

Applicant/permit holder	Status	Type of discharge	Impact level
CWDP			
OTMS Mogs Saldanha	CWDP lapsed	Hydrostatic testing	Low
ArcelorMittal Saldanha Steel	CWDP lapsed	Reverse Osmosis, Desalination (brine)	Low
Sea Harvest Corporation (Pty) Ltd	Permit granted	Fish processing effluent and brine	Medium
Transnet State Owned Company (SOC) Ltd	Permit granted	Desalination (brine)	Medium
Transnet Port Terminals	Permit granted	Industrial Storm Water	Medium
Sunrise Energy (Pty) Ltd	Permit lapsed	Once-off discharge (Hydrostatic testing)	Low
GDA			
Saldanha Lobster	GDA	Unknown (processing/ holding facility?)	Unknown
Saldanha Bay Oyster Company-Pepper Bay	GDA	Aquaculture	Not Reported
Jacobsbaai Sea Products	GDA	Aquaculture	Not Reported
Aqua Foods SA	GDA	Aquaculture	Not Reported
Oceangrown Aquaculture	GDA	Aquaculture	Not Reported
Saldanha Oyster	GDA	Holding facility	Low

4.10.2 SEWAGE AND ASSOCIATED WASTEWATERS

BACKGROUND

Until recently sewage discharge was arguably the most important waste product discharged into Saldanha Bay in terms of 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. It is important to note that the wastewater received by sewage treatment plants also contains other degradable organic wastes such as agricultural waste, food processing wastes (e.g. from fish factories and slaughterhouses), brewing and distillery wastes, paper pulp mill wastes, chemical industry wastes and oil spillages.

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 stormwater 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). 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 IOT and the causeway, as well as the multitude of organic pollution sources within this area (e.g. fish factories, mariculture farms, sewage overflow from pump stations, and stormwater 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. Generally, chlorine-containing biocides are 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. Should dosing not be undertaken correctly, discharged effluent may be high in chlorine (which, as a biocide is toxic to most organisms) as well as a range of toxic disinfection by-products through its reactions with organic compounds (Richardson et al. 2007, la Farré et al. 2008, Sedlak & Von Gunten 2011). While chlorine breaks down naturally through reaction with organic matter and in the presence of sunlight, the GDA limits specify that concentrations should not exceed 0.25 mg/l at the end of pipe (i.e. just before discharge).

There are two Wastewater Treatment Works (WWTWs) 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. Mechanical and electrical equipment upgrades to the pump stations in Saldanha and Langebaan to prevent overflow of raw sewage were completed in 2017 (Gavin Williams (SBM) *pers. comm.* 2016). There are also approximately 200 conservancy tanks in Langebaan, east of Club Mykonos (Elmi Pretorius (SBM) *pers. comm.* 2014). Overflow of these tanks is considered an unlikely event today, as the municipality empties these tanks on a regular basis (Gavin Williams (SBM) *pers. comm.* 2014).

REUSE OF TREATED WASTEWATER

The 2017/18 drought in the Western Cape 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 in Saldanha Bay. Water shortages must be understood as the 'new normal'—long-term climate models predict that global warming will result in drier conditions in the Western Cape. Growing demand for water by industry, especially in the SBM, must also be taken into account for future planning.

It is reported that no effluent from the Langebaan WWTW is entering the Bay, with all treated water diverted for alternate use. Likewise, there are several water users receiving treated effluent from the Saldanha WWTW, including the Weskus School (200 m^3/day), Saldanha Sports Ground (Stadium and practice field/350 m^3/day), Blue Bay Lodge (~70 m^3/day however, they tend to abstract more due to instream retention ponds). When operations, small intermittent amounts were used by Arcelor Mittal as required for dust suppression (Quintin Williams (SBM) *pers. comm.* 2023). The balance of treated effluent that is not used for irrigation is currently discharged into the Bok River and ultimately ends up in Small Bay and the ocean. It is worth noting that the exact quantity of effluent entering the Bok River is unknown, and although it has been indicated previously that a flow gauge will be installed no results from this have been made available.

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. Industry in Saldanha Bay 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 (RO) to suit the needs of industry. Considering ongoing water shortages and the environmental impacts associated with the discharge of WWTW effluent, this was conceived as an attractive opportunity but has not been developed further.

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, this WWTW also receives and treats industrial wastewater from a range of industries including Sea Harvest, Hoedjiesbaai Hotel, Protea Hotel, Bongolethu Fishing Enterprises, SA Lobster, Transnet Port Authority, Abattoir and Duferco (wastewater from Arcelor Mittal was also received until its closure in the first quarter of 2020). By-laws manage contributor effluent quality and volumes, with monitoring and requirements penalties for transgression (Gavin Williams *pers. comm*. 2018) and the plant operates under a GDA.

The plant capacity of the Saldanha Bay WWTW is 5 000 m^3/day to accommodate the projected increased wastewater production, and various other improvements to the plant were also implemented to ensure that the treated wastewater is of acceptable quality, especially with the establishment of the Saldanha Bay IDZ (the IDZ funded and managed this project) (Section 4.4.5). Originally, the Saldanha WWTW was issued a discharge exemption under the NWA section 21(f) and (g), provided that the effluent volume did not exceed 958 000 m^3 per year and that the water quality of the treated effluent was compliant with the General Discharge Limits under the NWA. With the gazetting of the new GDA legislation, as long as the discharge volume into the marine environment remains less than 2000 m^3/day, the plant now operates under a GDA. Using the supplied inflow data from 2024/25 and volumes allocated to other users, the calculated average daily discharge entering the Bok River is approximately 1 400 m^3/day, which falls within the allowable volumes (<2 000 m^3/day) of the GDA regulations.

Data on the monthly trends in the effluent produced by the Saldanha WWTW and provided by the SBM are presented below[4]. Concentrations of contaminants in the effluent for the Saldanha Bay WWTW are compared with the GDA limits. The average daily effluent volumes have increased steadily over time until about 2013, after which values fluctuate around the 'old' NWA limit of 2 625 m^3/day (Figure 4.14). Since November 2019, the average daily discharge volume has been non-compliant 26% of the time with this limit, and the plant capacity limit was exceeded in November 2019 (Figure 4.14). However, over the last year (2024/25), this average daily limit was not exceeded, and the discharge was 100% compliant (Figure 4.14).

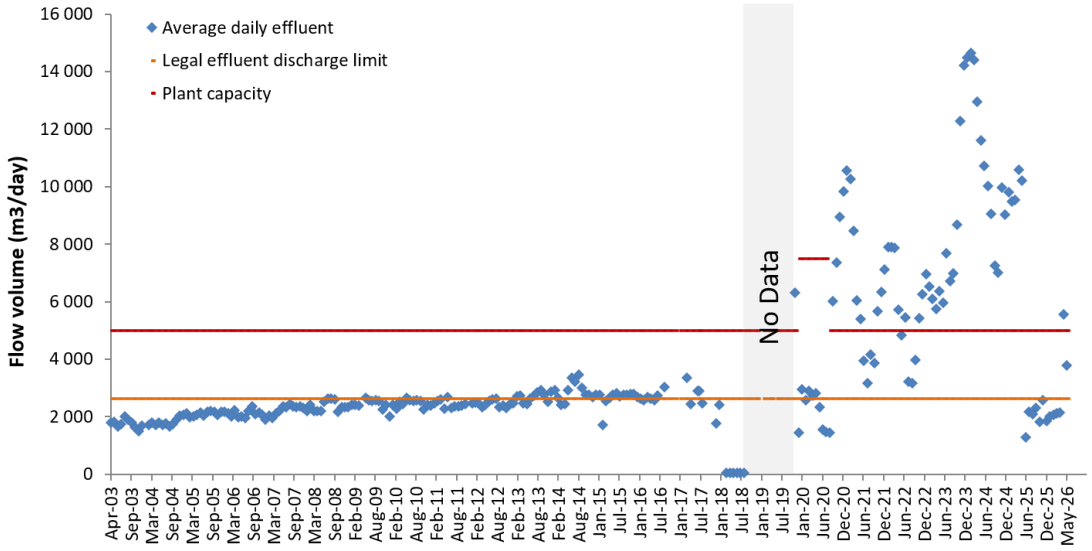


Figure 4.14. Average daily effluent (m^3/day) inflow for the Saldanha WWTW. The orange line indicates allowable discharge limits, while the design capacity of the plant is indicated by the red line. The red circle indicates times when no effluent was discharged into the Bay.

While a flow meter has reportedly been installed at the Bok River discharge point, it is not known whether the discharge volume is recorded. To ensure continued compliance with the discharge limit, it must be ensured that this flow gauge is operational, and that the data captured are collected and reported on as a continuous record of effluent volumes entering the Bay. Based on inflow data, the changes implemented by the SBM are positive and future interpretation of water quality results must continue to consider the volume of effluent entering the marine environment via the Bok River.

Concentrations of faecal coliforms in the effluent from the Saldanha WWTW exceeded the allowable limit of 1 000 org/100 ml on 56 occasions since 2003 (20% of the time) (Figure 4.15). A strong improvement is evident from November 2019, after which point the faecal coliform counts exceeded the limit ten times (15% non-compliance). The 2024/25, non-compliance was higher, at 67%, representing a worsening in compliance

[4] Water quality parameters recorded as "trace", "less than" or "greater than" were adjusted in accordance with the following standard international convention ("Trace" = half the detection limit; "Less than" = half the detection limit; and "Greater than" = detection limit multiplied by a factor of three.) It is important to note that these results are based on plant inflow volume data provided by the Saldanha Bay Municipality, as it is reported that, due its location within the plant, the outflow meter/gauge is affected by the gases in the inlet works and therefore continuously reports faulty readings.

but was improved on again in the most recent year, 2025/2026, with 25% non-compliance (Figure 4.15).

Since 2003, allowable limits for Total Suspended Solids (TSS) are exceeded 14% of the time (note that the limit was increased to 50 mg/l in 2022 under the GDA) (Figure 4.15). This increased limit has meant an improvement in compliance over time—since November 2019, TSS has only exceeded the limit once (35 mg/l in May 2021) (Figure 4.15).

Chemical Oxygen Demand (COD) is commonly used to indirectly measure the amount of organic material in water, and the filtered COD concentrations in the effluent exceeded the limit of 75 mg/l 19% of the time since 2003 (Figure 4.15). There has been significant improvement over time, with a strong compliance record in recent years. In the past year only September 2025 was non-compliant at 80.2mg/l, but it represents the first non-compliant results since June 2022 (Figure 4.15).

Nitrate-N levels in the effluent fluctuate over time, with an overall compliance of around 80% since 2003, but also with exceedances in 2005, 2009/2010, 2013, and 2016/2017 and 2021 (Figure 4.15). It is possible that generally higher Nitrate-N levels in 2017 can be attributed to more effective treatment of effluent in the new aeration basins, where more ammonia-N is converted into non-toxic nitrate-N by means of bacterial treatment processes. Conversely, low nitrate-N levels (<0.3 mg/l) between November 2017 and April 2019 complement extremely high levels of ammonia-N, indicating the lack of bacterial treatment. Since November 2019, there has been a 97% compliance rate, while in 2025/26, the nitrate-nitrogen levels in the Saldanha WWTW effluent were 100% compliant (Figure 4.15).

The concentration of orthophosphate in the effluent has only been measured since October 2007 and has an overall non-compliance rate per this period of 19% (Figure 4.15). There has been 100% compliance since the implementation of the 2022 GDA (Figure 4.15).

Ammonia-N (mg/l as N) levels have been flagged as of concern in the treated effluent in the past, with concentrations exceeding the limits some 50% of the time since April 2003 (Figure 4.15). The maximum concentrations were some 92 mg/l (October 2018), which is of concern, given that ammonia is toxic to aquatic organisms. It is notable, however, that effluent ammonia levels have improved significantly since November 2019, with a non-compliance rate of 38%. However, in 2025/26, effluent ammonia-N was non-compliant 66% of the time (Figure 4.15).

Permissible chlorine levels of 0.25 mg/l have been exceeded 63% of the time since 2003 (Figure 4.15). The situation has deteriorated further in the last year, with 92% non-compliance (11 of the 12 readings exceeded the permitted limits) (Figure 4.15). The 2022 GDA regulations do not provide values for free or total chlorine, the only chlorine limit provided is for total chlorine residual (0.2 mg/l) which is combined with free chlorine to provide total chlorine. Residual chlorine is not a value currently tested, and it may be worth altering the tests to include this parameter.

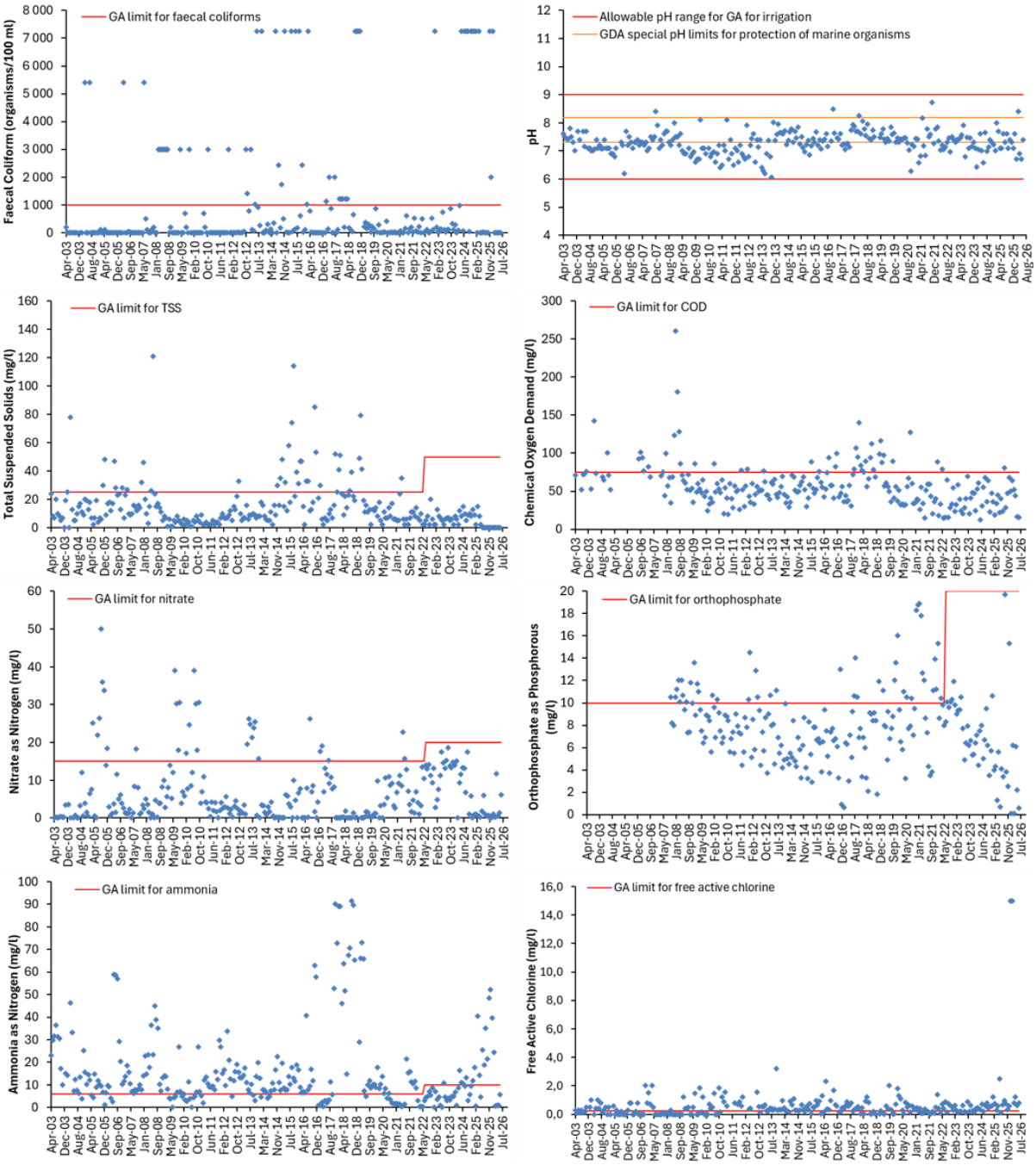


Figure 4.15. Saldanha WWTW effluent quality. The General Authorisation limits changed with the amended GDA gazetted in July 2022 are indicated by the red line.

LANGEBAAN WASTEWATER TREATMENT WORKS

While the Langebaan WWTW treats sewage by means of activated sludge with biological nutrient removal and drying ponds, the SBM favours alternative uses of wastewater from the Langebaan WWTW over discharge to the marine environment. In January 2018, SBM was successfully registered as a water user in terms of Section 22(1)(a)(iii). As such, the Langebaan WWTW is permitted to irrigate up to 73 000 m^3 (daily maximum of 200 m^3 per day) wastewater per annum on 12.68 ha. The majority of wastewater produced by the Langebaan WWTW is diverted to the Langebaan Country Estate for the irrigation of the golf course, with the remainder used to irrigate lawn on the WWTW premises as well as

the flower beds along Oostewal Road leading into Langebaan. The SBM is permitted to store treated effluent for irrigation purposes in ponds (maximum storage capacity of 4 485 m^3). Prior to irrigation, the wastewater is further treated by means of 11 polishing ponds. Currently, reports indicate that water restrictions and an agreement with the Langebaan Country estate to irrigate effluent within its boundaries during the winter months means that any discharge of effluent into the marine environment has been eliminated (Quintin Williams (SBM) *pers. comm.* 2023).

Various phased upgrades have been undertaken to improve the overall performance of the treatment plant (Gavin Williams (SBM) *pers. comm.* 2016). The ongoing phased approach and installation of new infrastructure will increase the capacity of the plant to 5–7 megalitre. However, over time, more effluent than currently absorbed by the Langebaan Country Club will be produced. Before this occurs the SBM will need to find alternate user options or design options for the discharge of excess effluent. For example, the municipality is planning to use excess effluent to irrigate the lawn at the Langebaan Sports Complex.

Water quality parameters presented here represent those measured at the end of pipe after treatment, but before storage and treatment in the 11 polishing ponds. Given the use of the effluent for irrigation, trends of water quality parameters for this end of pipe effluent at the Langebaan WWTW effluent between 2009 and 2026 have mostly been compared to GDA limits for irrigation (Table 4.4). However, due to possible overflow and occasional potential discharges of effluent into the marine environment, the effluent monitoring results are also compared to limits more relevant to the inshore marine environment (i.e. special limits specified as part of the 2022 GDA regulations).

Table 4.4. 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 or more than 9 pH units
Electrical conductivity	Not exceed 200 milliSiemens per metre (mS/m)
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

It is noteworthy that for quite some time, the amount of wastewater entering the marine environment has been negligible 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 2009–2017 (Figure 4.16). Overall, effluent volumes peak over the December holidays when plant capacity was often reached or exceeded (e.g. December 2016, average daily effluent volumes were 2 840 m^3). The legal limit for effluent production increased to 4 485 m^3/day in

January 2018 when the SBM was issued with a new GA permission. Since then, the Langebaan WWTW has been compliant in terms of the legal effluent volume limit (Figure 4.16). However, the hydraulic design capacity (3 500 m^3) was exceeded in February 2019 (119% capacity), in March 2020 (101%), in September 2024 (104%), January 2025 (106%), August 2025 (105%), January 2026 (112%), May 2026 (103%) and June 2026 (105%) (Figure 4.16).

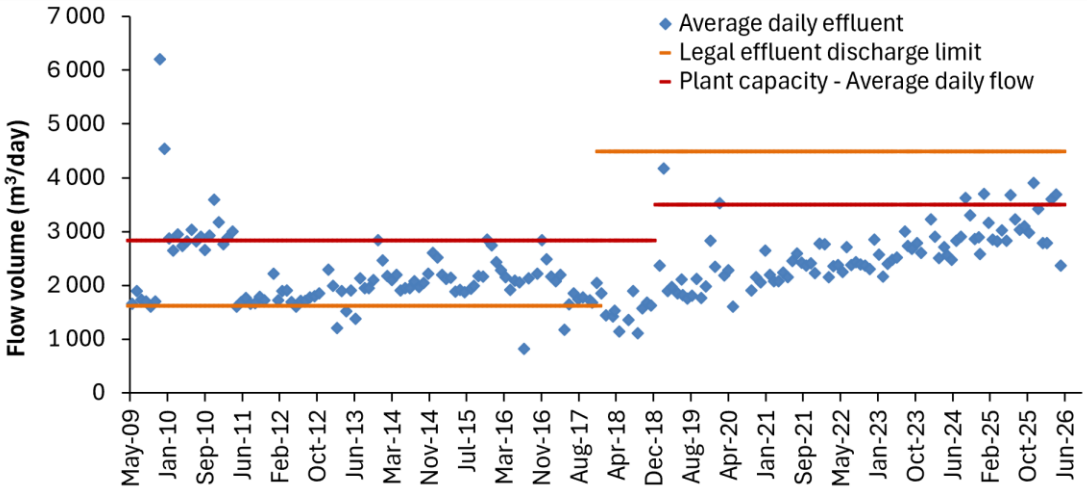


Figure 4.16. Average daily effluent volume (m^3/month) for Langebaan WWTW effluent. The orange line indicates that allowable discharge limits, while the design capacity of the plant is indicated by the red line.

Up until the start of 2024, the wastewater has consistently fallen within the pH range stipulated in the GA for the irrigation of <500 m^3 wastewater (Figure 4.17). Since January 2024 however, the pH has fallen outside of these limits nine times (non-compliance rate of 30%), including a highly acidic reading of 2.8 in February 2024, and a highly basic reading of 11.2 in June 2025 (Figure 4.17). The pH of the wastewater effluent has fallen outside of the more stringent range for the protection of the inshore marine environment limits some 46% of the time since 2009, with more basic (pH > 8.2) values recorded five times and more acidic (pH <7.3) values recorded on 88 occasions (Figure 4.17). There is evidence of a deterioration in this water quality parameter over time, and investigations should be made to ascertain and rectify the cause.

COD in filtered effluent fluctuates around the allowable limit for the protection of marine organisms (75 mg/l), exceeding the limit 24% of the time since June 2009 (Figure 4.17). Compliance does seem to have improved in recent years, with 83% compliance in 2025/26.

The concentrations of faecal coliforms in the effluent from the Langebaan WWTW have never exceeded the limit of 100 000 organisms/100 ml imposed by the GA applicable to irrigation; however, this limit is deemed unacceptable in terms of recreational and mariculture concerns (Figure 4.17). The GDA threshold of 1 000 organisms/100 ml has been exceeded 32% of the time since June 2009, and in the 2025/26 period, there was a compliance rate of 100% (Figure 4.17).

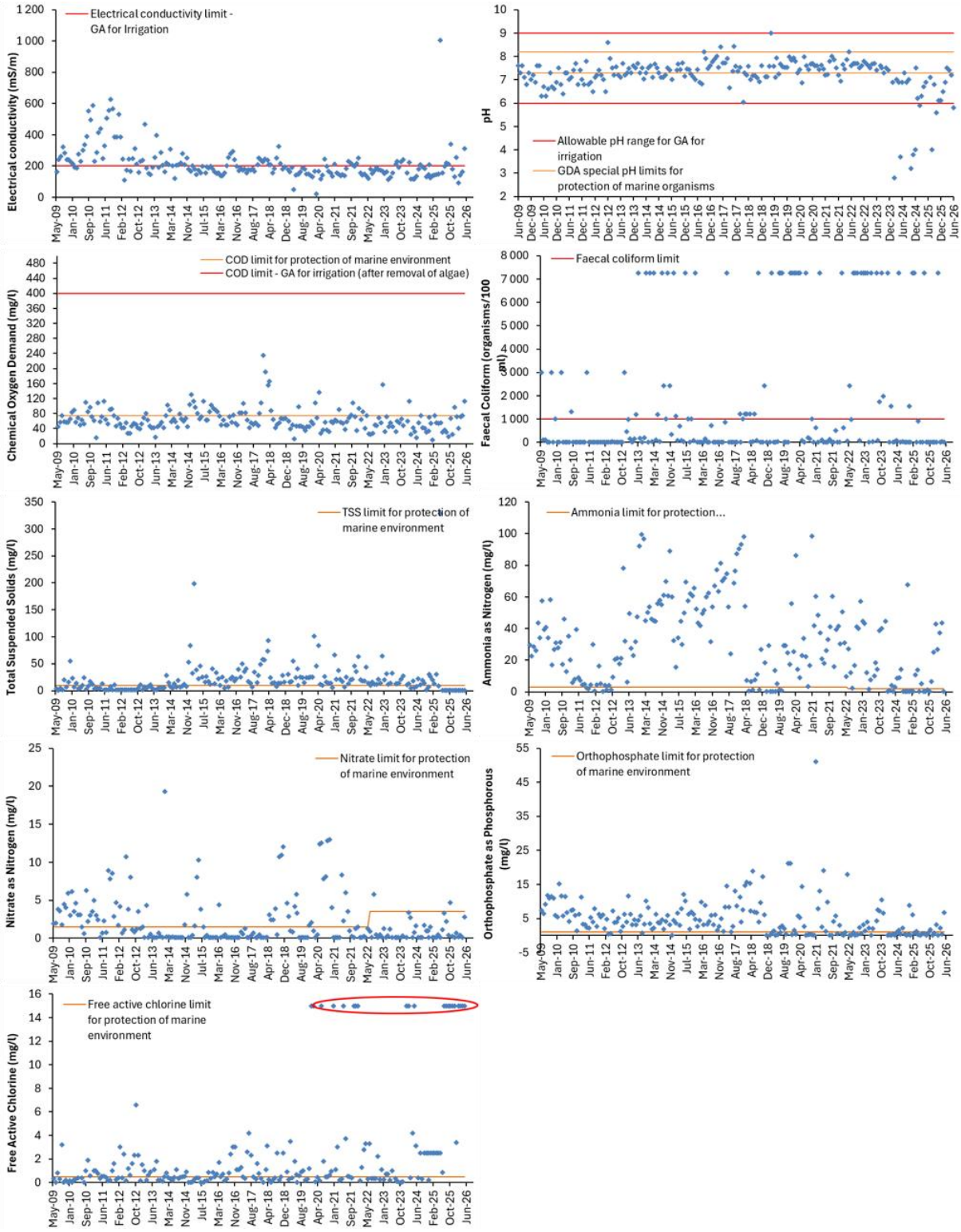


Figure 4.17. Langebaan WWTW effluent quality. The red lines indicate the limits as per the GA for irrigation, the orange line indicates the GDA special limit.

While no TSS limit is prescribed by the GA applicable for irrigation of wastewater, the water user is required to remove all suspended solids prior to irrigation of the wastewater. The polishing ponds on the Langebaan Country Estate act as settlement ponds and TSS is likely to be lower than shown in Figure 4.17. However, TSS values at the end of pipe at

the WWTW (shown here) exceeded the recommended special limit for the protection of the inshore marine environment on 128 occasions since June 2009 (62% of the time) (Figure 4.17). However, in 2024/25, the situation was worse, with a non-compliance rate of 75% with a maximum concentration of 329 mg/l reported in June 2025 (Figure 4.17). However, it seems that the TSS issue has been addressed as the most recent years compliance was 100%. It is evident that annual peaks in the concentration occur at the end of the summer or early autumn each year.

No ammonia-N limit is prescribed by the GA applicable to irrigation of wastewater. However, ammonia is very toxic to marine life, and the GDA special limit is set at 2 mg/l. While there is consistent non-compliance with these limits, with a non-compliance rate of 85% since 2009, there appears to have been some improvement over the last year, with a non-compliance rate of 58% in 2025/26 (Figure 4.17).

While nitrate-N is not considered to be directly toxic to marine life, it is a primary nutrient and could stimulate nuisance algae growth near the outfall point and its surrounds should any of the effluent reach the sea. While no nitrate-N limit is prescribed by the GA applicable to irrigation of wastewater, the GDA special limits is set at 3.5 mg/l. This limit has been exceeded on 42 occasions since June 2009 (20% of the time) with the 67% compliance in 2025/2026 (Figure 4.17).

Orthophosphate is usually not the limiting nutrient for primary production in the marine environment, and the recommended limit applicable for discharges into the inshore marine environment is 1 mg/l. While the orthophosphate concentration in the Langebaan WWTW effluent is considerably higher than 1 mg/l (74% non-compliance since June 2009), levels improved over time, with 33% non-compliance in 2025/26 (Figure 4.17).

While there is no free active chlorine limit prescribed by the GA applicable to irrigation of wastewater, the biocidal nature of chlorine means that it is highly toxic to marine life (the recommended limit to protect the inshore marine environment is 0.5 mg/l). Since 2009, there has been a 52% non-compliance with free chlorine levels in the effluent, and in 2024/25, the non-compliance worsened to 83% (Figure 4.17). These values are above the recommended limit for the safety of marine organisms i.e. these levels are significantly higher than what would be considered acceptable if discharged into the nearshore environment and more careful dosing of chlorine should be implemented.

4.10.3 Reverse Osmosis Plants

Reverse Osmosis (RO) 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. RO 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 WCDM and the rapidly developing industry in Saldanha Bay requires vast quantities of potable water for their operations. Construction of RO 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. Details on the three authorised desalinisation RO plants in the Saldanha Bay area are provided below:

- TNPA built a RO plant in Saldanha Bay to produce freshwater for dust mitigation during the loading and offloading of iron ore, and the plant was granted a CWDP in terms of ICMA in 2017 (DEA: O&C *pers. comm.* 2017).
- A marine baseline monitoring study was conducted prior to the commissioning of the RO plant (Hutchings & Clark 2011), and monitoring of the physical and chemical characteristics of the receiving environment were also conducted from June 2010 to March 2011 to establish a baseline prior to commencement of operation (Van Ballegooyen et al. 2012).
- Monitoring of the marine environment in fulfilment of the EMPr was conducted by the CSIR in 2014 and 2015. Cellozyme Environmental were appointed to undertake the monitoring in 2018 but all monitoring ceased shortly after this.
- The operational monitoring requirements as specified by the Water Use Licence and the Record of Decision issued by the former DEA for the RO plant included monthly monitoring of temperature, salinity, dissolved oxygen, turbidity, concentrations in the brine basin, continuous (hourly) monitoring of temperature, salinity, dissolved oxygen, and turbidity at representative outfall monitoring station and a reference station for at least one year; and surveys of trace metals and benthic macrofauna to be conducted bi-annually for an unspecified period.
- The WCDM has proposed the construction of an additional RO plant in the Saldanha Bay area as a long-term sustainable alternative water source 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).
- A 2007 feasibility study identified a 25 500 m^3/day sea water desalination plant as the most cost-effective solution. An EA was granted on 13 August 2013 for the development of an RO plant on the farm Klipdrift at Danger Bay on a portion of municipal owned land. Subsequent costs estimates, however, showed that the proposed desalination plant and bulk infrastructure would cost more than double the initial estimated cost. The project has been put on hold as the WCDM did not receive Grant funding for the construction of the desalination plant (David Wright (SBM) *pers. comm*. 2020). Should funds become available, construction of this RO plant is planned to be executed in three phases, with an initial capacity of 8.5 million litres later building up to a final capacity of 25.5 million litres.
- Alternatively, an application for additional allocation of water sourced from the Berg River was submitted by the WCDM. In October 2017, SBM received an increased allocation of 20 427 000 m^3 per annum from the WCWSS which equates to 56 megalitres/day and is higher than the current water demand. This in conjunction with the Langebaan Road Well field and Hopefield Well field development has reduced the need for the Desalination plant at this stage, however, SBM with retain the project in their future augmentation projects planning (David Wright (SBM) *pers. comm.* 2020).
- ArcelorMittal was a largely export-focussed steel plant close to the deep-sea Port of Saldanha that was closed in the first quarter of 2020. Prior to this, operations required approximately 6 500 m^3/day of freshwater, which represented approximately 25% of SBMs potable water total usage. ArcelorMittal Saldanha modified its water treatment infrastructure to partially replace its fresh water

supply with treated municipal sewage wastewater (from the Saldanha WWTW). Since the closure of the ArcelorMittal Saldanha Works, no industrial effluent has been sent to the Saldanha WWTW, and no treated effluent has been pumped from Saldanha WWTW to ArcelorMittal for use in the RO Plant. The plant's CWDP has lapsed.

- With the implementation of water restrictions in 2017, Sea Harvest implemented a RO plant (commissioned in April 2018) for the reclamation of potable water from seawater for daily operational consumption. This enabled the plant to reduce its freshwater consumption by 90% since 2016 (see Section 4.10.5 below). As of the end of July 2022, it is reported that 99.8% of all water consumed at the Saldanha Operations is sourced from the Desalination Plant.

4.10.4 STORMWATER

Stormwater 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. Stormwater running over these surfaces accumulates debris and chemical contaminants, which then enters water bodies untreated and may eventually lead to environmental degradation (Booth & Jackson 1997, Bay et al. 2003). 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).

Contaminants that are commonly introduced into coastal areas via stormwater 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) (see Table 4.5). The concentrations of these potentially toxic compounds may fall above the South African threshold guidelines for coastal and marine waters (Table 4.5). It is likely that introduction of contaminants via stormwater runoff negatively impact the health of the marine environment, especially during the "first flush" period as winter rains arrive. It is difficult to characterise and treat stormwater runoff prior to discharge 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 stormwater runoff is to target the source of the problem by finding ways that prevent contaminants from entering stormwater systems. This involves public education as well as effort from town planning and municipalities to implement stormwater management programmes.

Table 4.5. Typical concentrations of water quality constituents in stormwater runoff (residential and industrial) (from CSIR (2002)) and South African 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 Guideline limits
Total suspended solids (TSS) (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*

Parameter	Residential	Industrial	Water Quality Guideline limits
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**

All residential and industrial stormwater outlets drain into the sea. The volume of stormwater runoff entering waterways is directly related to the catchment characteristics and rainfall (Table 4.6 and Figure 4.18). Stormwater runoff is highly seasonal and peaks in the wet months of May–August (Figure 4.18). Due to the rapid pace of holiday and retail development in the area, Langebaan residential area produces the greatest volumes of stormwater runoff, followed by the industrial areas, with lower volumes arising from the Saldanha residential area (Figure 4.18). The actual load of pollutants entering the Bay and Lagoon via this stormwater can only be accurately estimated when measurements of stormwater contaminants in the stormwater systems of these areas are made.

Stormwater 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 4.18).

Table 4.6. Monthly rainfall data (mm) for Saldanha Bay over the period 1895–1999 (Visser et al. 2007).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Mean annual precipitation	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

Historically, stormwater from the Port of Saldanha and IOT was allowed to overflow into the Bay but most of this is now diverted to stormwater evaporation ponds and any material settling in these ponds is trucked to a landfill site.

A Stormwater Management Master Plan was drafted and is amended as new issues arise (living document) (SBM 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, with three additional large ponds proposed (note that these numbers may change as the Stormwater Master Plan is amended).

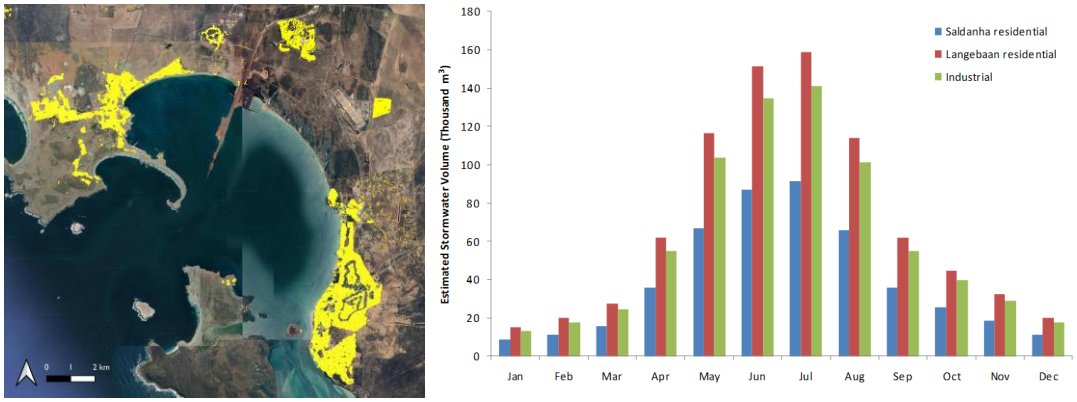


Figure 4.18. (Left) Spatial extent of residential and industrial areas (in yellow) surrounding Saldanha Bay and Langebaan Lagoon from which stormwater runoff is likely to enter the sea. (Right) Monthly estimated stormwater volume (m^3) from data in Table 4.6 above. Note that runoff from the Port of Saldanha and ore terminal have been excluded as this is now reportedly all diverted to stormwater evaporation ponds.

There are approximately 15 stormwater outlets in the Saldanha Bay residential area. A Basic Assessment Process for the upgrade the existing stormwater infrastructure in the operational and non-operational areas within the boundaries of the Port of Saldanha was published in 2019. These upgrades include the development of three new stormwater retention ponds, the expansion and reshaping of existing stormwater retention ponds, development of a wastewater treatment facility, and upgrading of the stormwater management infrastructure as well as maintenance of existing ones. The process is still ongoing.

There are about 20 outlets for stormwater that drain directly into the Langebaan Lagoon. Concerns and complaints around poor stormwater management have been publicly raised by the residents of 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. Following recent heavy rains in the winter of 2024 stormwater runoff collected on roads before washing into the Bay via Leentjiesklip beach. Other concerns registered by the SBM include deterioration/destructions of wetlands and canalisation of streams and rivers reducing the assimilative and dissipative capacity of the natural environment, inadequate capacity of stormwater retention facilities east of Oostewal Street, the impact of stormwater effluent containing pollutants from roads, private properties and businesses discharging into the Langebaan Lagoon, the lack of maintenance of conveyance systems with large sediment deposits and the impact on tourism market due to deteriorating aesthetic value.

Despite efforts by the iron ore industry to reduce dust (refer to Section 4.3.3) and to divert stormwater to evaporation ponds, Saldanha Bay experiences frequent and considerable pollution, especially when the terminals are washed down (Figure 4.19). Iron in the solid state affects organism by either smothering or through physical damage, thereby reducing the survival fitness of the affected organism (Erasmus & De Villiers 1982, Woolsey & Wilkinson 2007, Anchor 2012). However, if iron is dissolved through chemical reactions with organic matter and oxygen, it becomes available to organisms in the marine environment as a micronutrient blooms (Anchor 2012). Toxin levels in

phytoplankton responsible for red tides also increase as a response to enhanced dissolved iron levels (He et al. 2010). 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 4.19. Pollution of Saldanha Bay by particulate iron carried by stormwater runoff (Source: Jaco Kotze, September 2014, Langebaan Rate Payers Association).

4.10.5 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 (Figure 4.20). Premier Fishing is currently in the process of re-commissioning and upgrading their fish processing plant.

SA Lobster Exporters (Oceana Lobster Saldanha) discharges 40 000 to 60 000 m^3 seawater from their operations into Pepper Bay. 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. However, it is listed as operating under a GDA.

Sea Harvest is a predominantly demersal trawl fishing company with a fish processing plant situated near the base of the causeway to Marcus Island in Saldanha Bay. The plant processes mostly hake (*Merluccius paradoxus* and *M. capensis*) into a variety of primary fish products including fillets, cutlets, steaks and loins. As part of its operations, Sea Harvest discharges large volumes of treated effluent from the fish processing plant to the sea. An RO plant was commissioned at the site in 2018 in response to municipal water restrictions (Section 4.10.3). The final effluent discharges along the Government Jetty (Figure 4.21) and comprises brine, Fresh Fish Processing factory effluent (i.e. process seawater used for cleaning processes to save potable water) and treated Added Value factory effluent. An amended CWDP was issued by the DEA:O&C on 7 November 2019. A monitoring programme for this outfall was undertaken by Anchor from 2019–2021 as per the CWDP conditions. An in-situ water sampling instrument was moored 100 m from the outfall to assess compliance with the CWDP limits for temperature and salinity, and to validate the results of the dispersion modelling study that was undertaken as part of the CWDP amendment process. In addition, quarterly water quality monitoring and two annual benthic assessments (sediment quality and benthic macrofauna community assessments) were conducted. Sea Harvest is committed to meeting effluent quality

thresholds and environmental monitoring requirements as stipulated in the CWDP and continuous investigations are conducted in order to improve effluent quality.

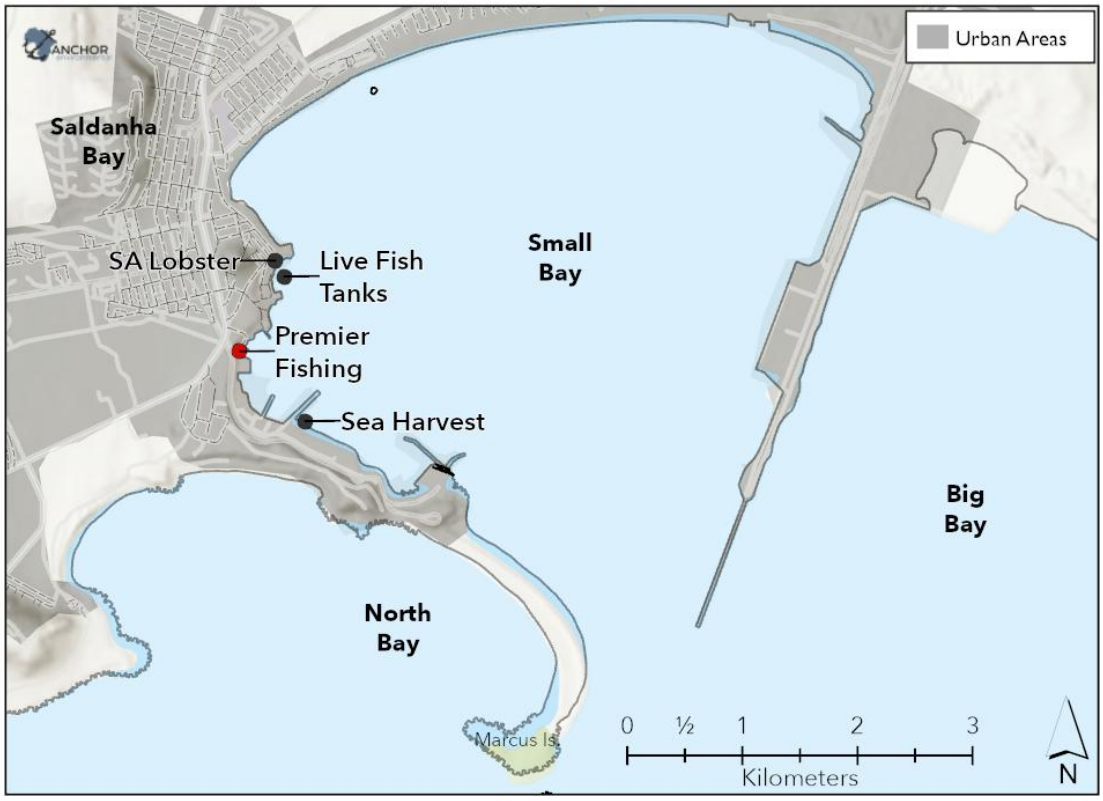


Figure 4.20. Location of seawater intakes and discharges for current and proposed seafood processing factories in Saldanha Bay. Current factories are indicated in black while the proposed Premier Fishing Fish Processing Plant is indicated in red.

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. An EA was granted in June 2013, and an Atmospheric Emission Licence was approved in April 2014 (this has been appealed). An application for a CWDP in terms of ICMA was submitted to the Competent Authority (the then DEA:O&C) for the discharge of cooling water containing condensate from the plant's scrubber to the sea, and 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, the DEA:O&C requested additional information, and that the application is subjected to another round of public participation. No CWDP has since been issued, and construction/operation has not commenced (former DEA:O&C 2017). Sea Harvest remains the only Fish Processing Plant to be granted a CWDP (DFFE 2024).

Figure 4.21. 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.

4.11 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).

The white stumpnose *Rhabdosargus globiceps* stock in the Bay suffered severely as a result of recruitment overfishing over the last two decades (too many fish being caught before they had a chance to spawn) but reduction in fishing effort during the COVID-19 pandemic seems to have allowed some fish to spawn successfully, evidenced by an increase in the numbers of new recruits in the Bay in recent years (white stumpnose remains of major concern although recruitment improved further in 2026, to 44.8 fish/haul). This recovery is likely to be short-lived though unless levels of fishing mortality are

reduced through the implementation of more conservative catch limits (reduced bag limits and increased minimum size limits).

The reported annual catch of harders declined from around 130 tonnes per year over the period 2008–2012 to about 90 tonnes per year over the period 2013–2016, whilst effort remained fairly constant (Horton et al. 2019) (Figure 4.22). Data on fishing effort in recent years are lacking unfortunately.

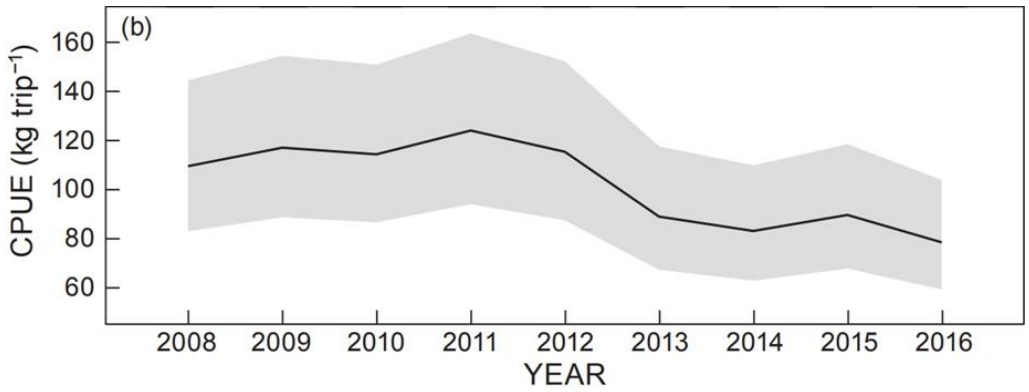


Figure 4.22. The standardised catch-per-unit-effort (CPUE) estimates for harders in the commercial net fishery at Saldanha Bay and Langebaan Lagoon (with 95% confidence intervals, grey area) derived from mandatory catch records kept between 2008 and 2016. Source: Horton et al (2019).

4.12 MARINE AQUACULTURE

4.12.1 OVERVIEW

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 since the 1980s. The development of the aquaculture sector was identified as a key priority of Operation Phakisa as a means of job creation and the local economy (Section 4.9). Under the Operation Phakisa Delivery Unit, the then DAFF applied for and was granted an EA in January 2018 for the establishment of a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay. The latest EA amendment was issued on 2 May 2024 (14/12/16/3/3/1/1728/AM3). The 2018 EA permitted the expansion of the total area available for aquaculture in the Bay to a maximum area of 884 ha within four precincts (Small Bay, Big Bay, Outer Bay North and Outer Bay South) (Figure 4.23). As of September 2025, there are 30 total allocated rights, with 24 operational (DFFE *pers. comm.* 2025).

As part of the EA and EMPr conditions, the DFFE: Branch Fisheries Management appointed an Environmental Control Officer to oversee the construction and operational phase of the ADZ and set up two committees: (1) a Consultative Forum (public and industry forum) which meets every three months, and (2) the Aquaculture Management Committee (government committee) which meets every two months, to ensure that the implementation of the ADZ is in line with the requirements specified in the EA and EMPr. Infrastructure guidelines for were also published and distributed to operators for each farm providing a specified number of mussel lines, oyster lines and/or mussel rafts.

Production was increased to 15 000 tpa ungraded shellfish (7 500 tpa graded) by the AMC on 24 November 2022. This adaptive environmental management was informed by environmental monitoring. Historical monitoring work included the development of a Sampling Plan for the ADZ, a dispersion model for the finfish farming and baseline macrofauna survey sampling. In 2020, the Branch Fisheries Management appointed Anchor to compile the ADZ baseline data collected in 2018 into a Benthic Survey Report (Mostert et al. 2020b) and to conduct the 2020 Annual Redox Survey and compile the resulting report (Mostert et al. 2020a). Additionally, the WWF South Africa through its Fish for Good initiative is currently implementing a Fisheries Improvement Project with the Saldanha Bay mussel sector (which is designated as a "catch and grow" fishery by the Marine Stewardship Council). WWF (SA) appointed Anchor to undertake the 2021 Benthic monitoring survey (Dawson et al. 2021) and conduct the annual benthic chemical surveys for the Saldanha Bay ADZ for 2021, 2022, and 2023 (Gihwala et al. 2021, 2022, Dawson et al. 2023) to fulfilling the Sampling Plan requirements.

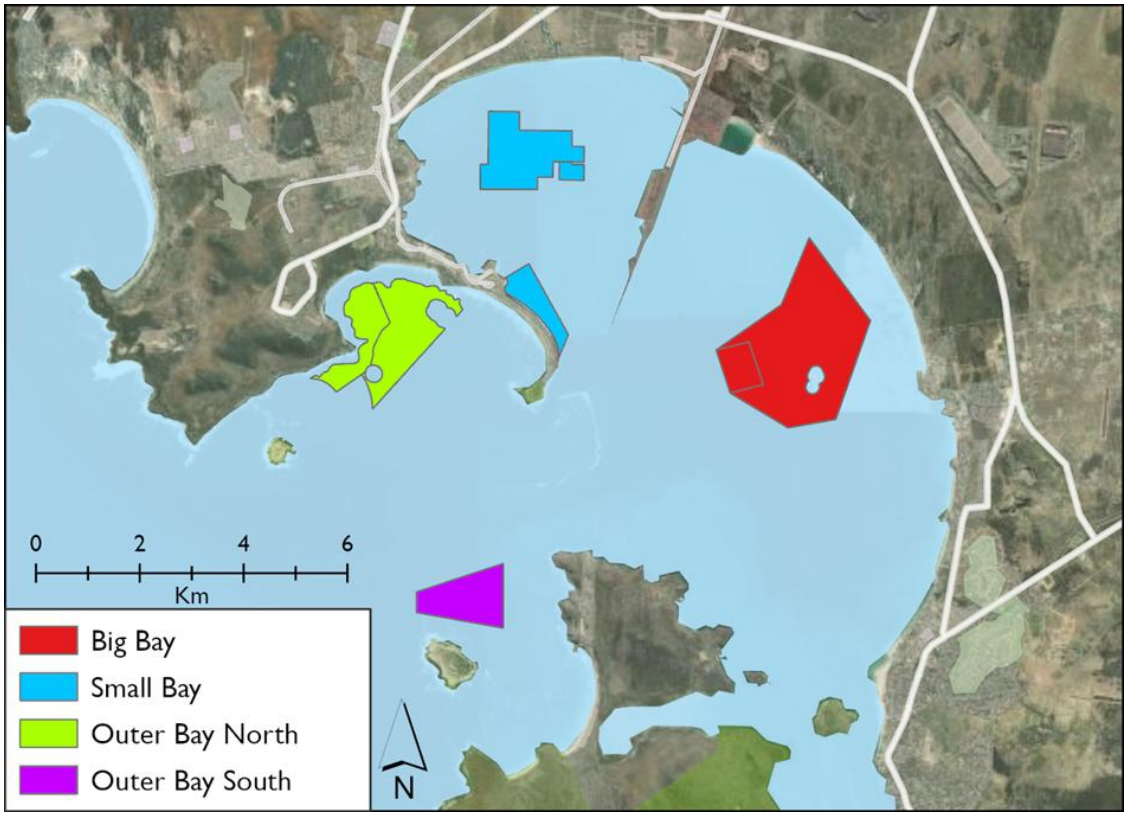


Figure 4.23. The four precincts that make up the entire ADZ area (884 ha).

Further work includes the ongoing maintenance of continuous monitoring sensors within the Bay, a survey of the hard substrate/reef structure in Big Bay (Dawson et al. 2022), data collection for dispersion model validation and a report on the first qualitative sampling of fouling organisms (the latter has not yet been released to the public and is therefore not summarised here). Additionally, reports have been issued to the DFFE on the scientific recommendations for the management and expansion of the ADZ along with updates to the Sampling Plan, based on the results of monitoring work completed to date. Anchor Research and Monitoring were appointed by the DFFE to continue to conduct specialist monitoring within the ADZ precincts in accordance with the Sampling Plan. This includes

the completion of annual chemical surveys (the first of which was completed, and the report submitted in June 2024) and benthic macrofaunal monitoring (sampling was undertaken in March 2024 and the report was submitted in September 2024).

Various guidelines and protocols have been developed for managing the ADZ, including an ADZ Entanglement Guideline (May 2020), Compliance Strategy (June 2020) and Incident and Emergency Response Protocol (March 2021, revised November 2023) and an Operational and Management guideline (November 2020). The ADZ has had five external Compliance audits of the EA and EMPr conditions, which have resulted in no significant non-compliances. The ADZ is also audited monthly by the ADZ Environmental Control Officer to manage the compliance at both the farm and ADZ level. The mussel industry is in the process of undertaking a Fisheries Improvement programme to improve overall fishing practices in the Rope Grown Mussel Industry, enhance the management of the fishery, establish critical partnerships, generate community support to inspire change in other fisheries in South Africa, and improve the accessibility of the Marine Stewardship Council standard in South Africa and other countries in the global South Africa. This initiative also includes a farmer awareness training programme on Endangered Threatened and Protected Species as well as training on the EA and EMPr conditions.

Most established operators hold rights to farm mussels *Mytilus galloprovincialis* and *Choromytilus meridionalis* and the pacific oyster *Crassostrea gigas*, while finfish rights *Salmo salar* and *Oncorhynchus mykiss* have only been issued to two farms since 2014. There are experimental seaweed projects underway. While rights are in place, abalone, scallops, red bait and seaweed are currently not cultured. 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 Branch Fisheries Management therefore included alien trout species in their application and consequently, the 2018 EA includes alien finfish Atlantic salmon (*S. salar*, Coho salmon *O. kisutch*, king/Chinook salmon *O. tshawytscha*, rainbow trout *O. mykiss* and brown trout *S. 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. 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 yellowtail *Seriola lalandi*.

4.12.2 BIVALVE FARMING

Raft culture of mussels has taken place in Saldanha Bay since 1985 (Stenton-Dozey et al. 2001). Larvae of the mussels *M. galloprovincialis* and *C. meridionalis* attach themselves to ropes hanging from rafts and are harvested when mature. Mussels are graded, washed and harvested on board a boat.

Mussel production was fairly consistent between 2007 and 2011, after which it showed a steady increase, more than tripling from 2012 to 2019 (Figure 4.24). In 2020, mussel production dropped by roughly 25% to 2 276 tonnes, with the possibility that the COVID-19 pandemic influenced the production, as the highest production values on record were

reported in 2021 (3 459 tonnes) with the 2022 production dropping yearly until reaching 2048 tonnes in 2024 after which recovering to 2760 tonnes in 2025 (Figure 4.24).

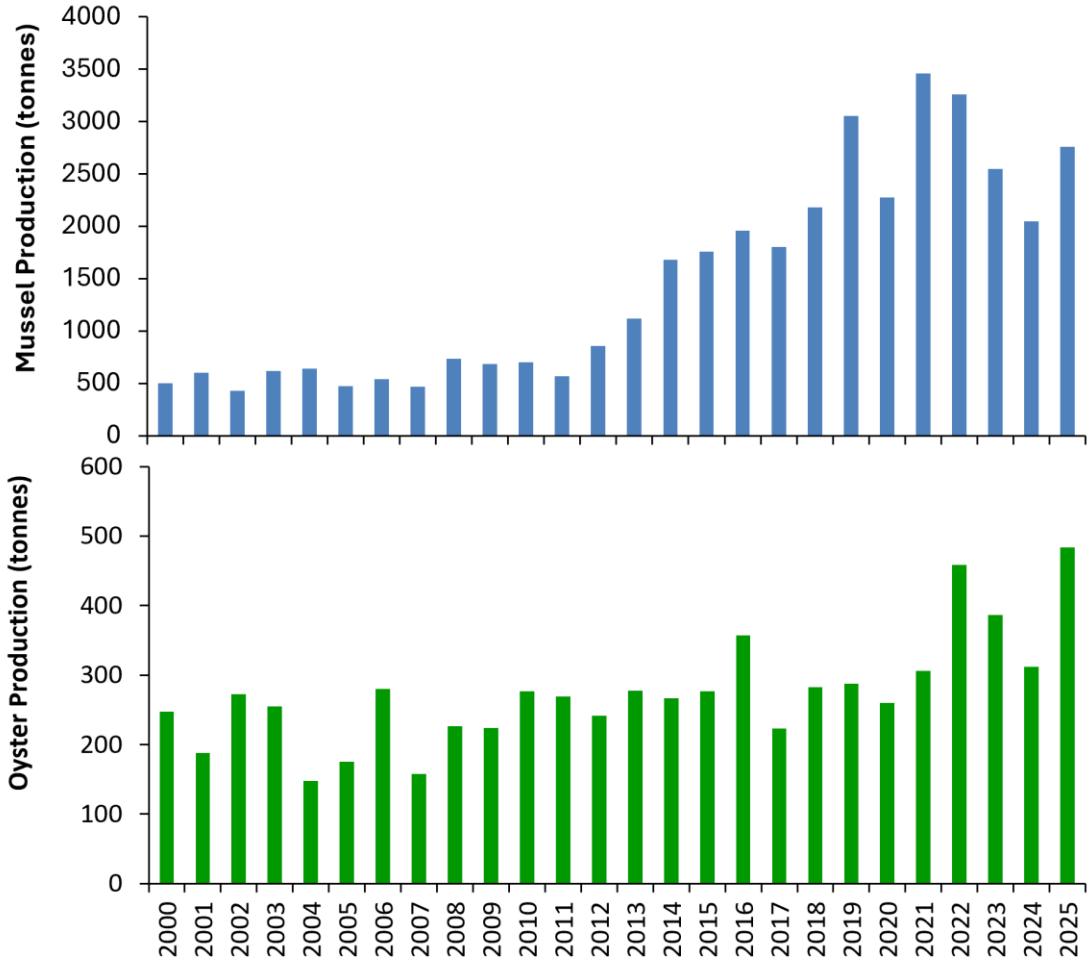


Figure 4.24. Annual mussel (top) and oyster (bottom) production (tonnes) in Saldanha Bay between 2000 and 2023 (source: DFFE 2023 unpublished data, which may be subject to change). Data represent production from January to December of each year.

Oyster production fluctuated around an average of 240 ± 45 tpa between 2000 and 2015. Oyster production peaked in 2016 at 357 tpa before decreasing again with the 2017 to 2021 average production being at 271 ± 32 tonnes of oysters (Figure 4.24). In 2022, the production reached an all-time record high of 458.9 tonnes, however, similar to mussel production it dropped until reaching 311 tonnes in 2024 and recovered to 483 tonnes in 2025.

The culture of mussels in Saldanha Bay may create organic enrichment and anoxia in sediments under mussel rafts (Stenton-Dozey et al. 2001). The ratios of carbon to nitrogen indicated that the source of the contamination was mainly pseudofaeces, decaying mussels and fouling species. In addition, it was found that the biomass of macrofauna was reduced under the rafts and the community structure and composition had been altered (Stenton-Dozey et al. 2001). Previous work has demonstrated that, based on the ecological carrying capacity of Saldanha Bay with regards to bivalve (in particular mussels and oysters) farming, the sector could increase 10 to 28-fold,

potentially creating an additional 940 to 2 500 jobs for the region without compromising the environment (Olivier et al. 2013). More recent work on the effects of mussel farming on benthic biogeochemistry and macrofaunal communities has shown that species diversity was significantly lower in the farmed sites relative to control sites, and that variations in community structure between Small Bay and Big Bay was partially related to physical aspects (sediment particle size distribution and wave energy) (Probyn et al. 2023). Other results suggest that the longer a farm is in place the more the community may start to diverge/change to withstand disturbance, with suspension feeders the more important discriminating species for control sites, and detritivores and scavengers for farm sites; although exceptions were common (Probyn et al. 2023).

4.12.3 FINFISH CAGE FARMING

Saldanha Bay has been identified as one of very few areas where finfish cages can be installed successfully, and indeed finfish cage culture focused on the farming of salmonid species, (including Atlantic salmon and rainbow trout) has been pioneered in Saldanha Bay by Molapong Aquaculture (Pty) Ltd (Ecosense CC 2017). The success of the pilot phase (under 50 tonnes of finfish per annum in Big Bay) resulted in the application for and awarding of an EA (on 8 January 2018) for the phased installation of sea cages on 40 ha in Big Bay and 15 ha near Jutten Island for the production of finfish, mussels and seaweed in Saldanha Bay up to a total of 2 000 tonnes per year over both lease areas. However, after the conclusion of the pilot phase in 2021/2022, all finfish cages were removed and there is presently no active finfish mariculture in the Saldanha Bay ADZ and none planned for the foreseeable future. This is likely due to the fact that it has been determined that the majority of the sea floor below the designated Finfish area is covered by reef (~79.9%) and therefore development of the finfish area within Big Bay is not advised (Dawson et al. 2022).

4.12.4 KELP CULTIVATION

A pre-feasibility study investigating the potential for commercial kelp cultivation along the West Coast was published in 2022 as a collaborative effort between the aquaculture industry, government, research institutions and academic institutions, aimed to review and summarise available biological, environmental, and economic information on kelp culturing and to set up facilities for conducting pilot experiments on kelp seeding (CSIR 2022). The successful commercial cultivation of three West Coast kelp species (sea bamboo *Ecklonia maxima*, bladder kelp *Macrocystis pyrifera* and split fan kelp *Laminaria pallida*) would provide access to the global market given that South Africa already has an existing industry for kelp, based on the harvesting of natural populations and the collection of drift kelp washed up on the shoreline. Overall, it was recommended that investigations proceed to the techno-economic feasibility stage to facilitate the collection of further data and information to support a decision to invest in kelp production mariculture. Proposed pilot studies include the installation of 4 ha of kelp longlines within the Bay (to be harvested twice annually), and a rope raft for the cultivation of kelp in the ADZ.

4.12.5 MONITORING

ANNUAL BENTHIC CHEMICAL SURVEYS

Following international best practice, monitoring of benthic impacts below mariculture installations in Saldanha Bay is undertaken by DFFE to validate dispersion model predictions of impact. Organic deposition and the subsequent decomposition by sediment bacteria increases oxygen demand which can lead to anaerobic (without air) conditions in the seabed beneath both finfish and shellfish farms. Ammonification and sulphate reduction to sulphides occur as typical responses to lowering of the oxygen reduction (Redox) potential. Sediment organic carbon, redox potential (Eh) and total sulphides (S^{2-}) have effectively been used in describing adverse impacts below finfish aquaculture and can be used to classify sediments associated with fish farming into five organic enrichment groups: two oxic (oxygen present), two hypoxic (oxygen deprived), and one anoxic (without oxygen) group. The Aquaculture Stewardship Council provides threshold limits directly below the aquaculture structures, as well as at the edge of the Acceptable Zone of Effect (AZE), a prescribed distance from the aquaculture structures, defined as 30 m from a fish cage array or shellfish longlines (Aquaculture Stewardship Council 2017). Failure to meet the prescribed thresholds at the AZE limit or at finfish cages or directly below shellfish longlines will require management intervention and/or additional sampling (DAFF 2018). Non-compliance is dependent on the farm or AZE station being significantly greater than levels measured at the reference/control stations. Results for the chemical surveys are available for the period 2019–2025.

The 2025 annual redox survey of the Saldanha Bay ADZ was conducted during the annual Saldanha State of the Bay survey (April 2025), but these results are not as yet publicly available. Therefore the results of the 2024 survey are presented here (Dawson et al. 2024). Sediment samples for the measurement of redox potential and sulphide (S^{2-}) were collected at 38 stations in Big Bay, Small Bay and Outer Bay North (Figure 4.25).

On balance, the currently available annual chemical survey data suggested that current levels of mariculture production within the ADZ are not having significant negative impacts on sediment (and sediment water interchange) quality. It must be noted that the impact sampling sites are located in between aquaculture infrastructure and are not directly under bivalve longlines and rafts. These results do not, therefore, contradict the earlier studies where a localised impact directly beneath the rafts was detected. Small Bay is identified as the area with the highest average sediment sulphide concentrations which is attributed to both hydrodynamic processes, natural and anthropogenic inputs (including mariculture). Average redox concentrations for Small Bay were notably higher than those measured in sediments collected from Big Bay or Outer Bay North. This is attributed to both the regular, seasonal hypoxia of near bottom water due to upwelling linked water movements, organic loading and relatively high retention times; as well as anthropogenic carbon and nitrogen inputs including fish factory wastes, biogenic waste from mussel and oyster culture and sewage effluent (Clark et al. 2021).

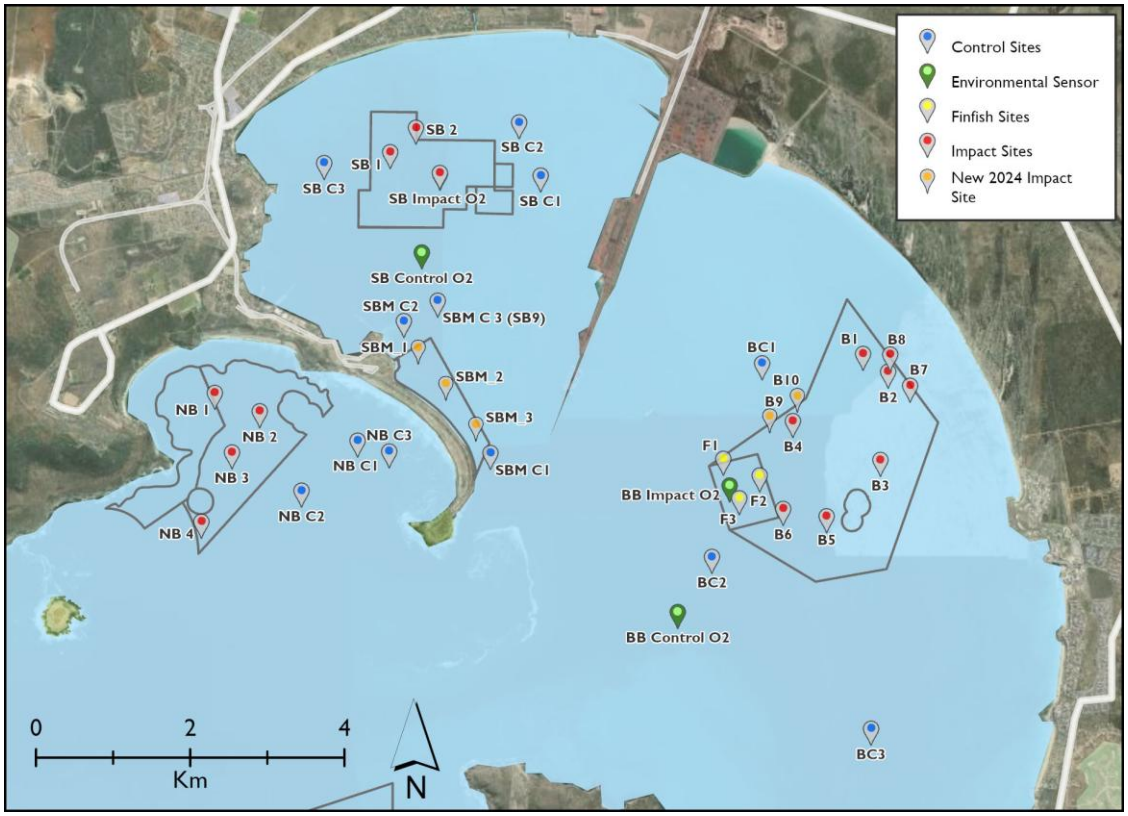


Figure 4.25. Stations sampled during the annual redox and ecological surveys of the Saldanha ADZ.

ECOLOGICAL MONITORING

The ADZ sampling plan requires a full benthic macrofauna survey to be undertaken every 3–5 years, with surveys to date undertaken in 2019 (the baseline), 2021 and 2024. Each survey includes the collection of benthic macrofauna samples and sediment samples for the analyses of Total Organic Carbon (TOC), Total Organic Nitrogen (TON) and particle size distribution. Additionally, trace metals samples are collected in the finfish precinct. The results of the latest ecological monitoring survey undertaken in March 2024 are summarised here (Dawson et al. 2024) (Figure 4.25). Data from samples collected at the impact sites within the ADZ boundaries, and reference sites falling outside the ADZ boundaries, are assessed and compared to the results of the 2019 baseline survey results to determine whether the aquaculture operations in these areas are having measurable benthic impacts. In addition, impact and reference sites in Small Bay and Small Bay mussel rafts were also sampled with the results of these compared to the Annual state of the Bay 2024 survey data within Small Bay.

Generally, the results show little significant differences between treatments (Impact/Reference sites) within the precincts in a given year, as well as limited variation across sampling years (2019 baseline or the 2021 and 2024 monitoring surveys), the only exception being that of the Impact Sites under the mussel rafts in Small Bay (Dawson et al. 2024). A number of monitoring parameters, including TOC and TON, sediment chemistry (sulphide and redox values taken from the annual chemical survey (Dawson et al. 2024), sediment Particle Size distribution, and community indices: Infaunal Trophic Index (ITI) and the AZTI Marine Biotic Index (AMBI), indicate that sites under the Small Bay mussel rafts are negatively impacted as a result of the long history of bivalve culture

(Dawson et al. 2024). Given that these sites are located in the oldest active aquaculture area it is possible that the impacts are cumulative. Therefore, it is suggested that some of rafts should be allowed to 'lie fallow' without active farming on a rotational basis. Below follows a summary of key findings:

- These results of Cumulative Abundance-Biomass plots of macrobenthic communities (Warwick 1993) highlight low to moderate levels of disturbance in Outer Bay North, low levels to no disturbance in Big Bay, with the exception of fairly high natural "disturbance" attributed to wave exposure and sediment movement at the finfish sites. The Small Bay mussel raft sites also display a low to moderate level of disturbance, while the rest of Small Bay shows no disturbance.
- In Outer Bay North, one site was 'moderately disturbed', and all other sites (impact and reference) exhibited a 'slightly disturbed' score. In addition, results showed that all communities within Outer Bay are experiencing low levels of disturbance, which suggests that exposure to wave action is an important driver of microbenthic community structure.
- Substantial variation in sediment particle size composition, TOC and TON concentrations at Small Bay mussel raft Impact Sites suggest a level of negative impact at these sites. Results for two of three sites indicate that the area is experiencing disturbance/negative effects as a result of the aquaculture. It is likely that the mussel raft area, which is the longest standing, actively farmed area, is experiencing the cumulative effects and may require management actions to prevent further accumulation of organic material under these rafts.
- Comparisons of the AMBI index across the three sampling campaigns suggest that the level of disturbance across the Bay has increased with time. This highlights to possible risk of long-term impacts, supporting the suggestion that although communities may not show impacts in the short term, extended exposure to aquaculture may lead to negative impacts on communities within the bay.

Servicing mussel ropes

5 GROUNDWATER

5.1 INTRODUCTION

The Greater Saldanha Bay region continues to grow steadily across multiple sectors, driven by its proximity to a deep-water port, suitability for industrial development, established infrastructure, and unique natural setting. These combined attributes make the area attractive for both economic activity and tourism, reinforcing the need for careful, long-term management that balances the interests of diverse stakeholders. Although the region receives limited freshwater inflow from surface water systems, groundwater remains a critical contributor to the Bay and Langebaan Lagoon. This input helps sustain marsh ecosystems, particularly around the Lagoon, and plays a key role in preventing hypersaline conditions during the dry summer months. Two main aquifer systems, the Langebaan Road Aquifer System and the Elandsfontein Aquifer System, provide this groundwater discharge via deep, interconnected paleochannels.

This chapter provides a summary of insights gained from the Saldanha Bay Municipality and Department of Water & Sanitation (DWS) groundwater-level and quality (electrical conductivity) monitoring networks in the Greater Saldanha Bay region over the past year. It also reviews groundwater-level and electrical-conductivity monitoring along a cross-section through the Elandsfontein mine, which sits on the paleochannel forming the groundwater pathway to the Langebaan Lagoon. The mine operates a closed-loop dewatering and managed aquifer recharge system, dewatering the pit and reinjecting that water down-gradient; this chapter presents current evidence on how well that system is maintaining pre-mining water levels and water quality along the pathway to the Lagoon.

Rainfall trends for the period 2017–2026 (to date) are presented in Figure 5.1, based on data obtained from the weather station based at the Elandsfontein Phosphate Mine. Rainfall volumes have increased since the 2015–2019 drought, with the last four years recording above-average totals. The 2023 annual rainfall (584 mm) was the highest over the past 8 years, while January to July 2026 rainfall volumes have been recorded at 263 mm.

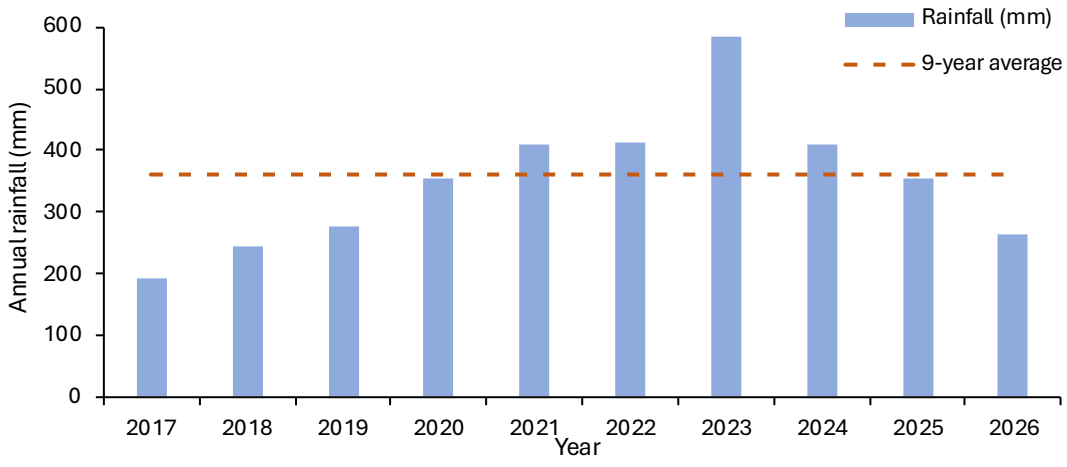


Figure 5.1. Annual rainfall (2017–present) obtained from the weather station at the Elandsfontein Phosphate Mine. 2026 data covers January–July.

The underlying hydrogeological context has not changed significantly since previous reports; readers seeking a detailed background can refer to the 2021 SOB report (Clark et al. 2021) for an in-depth description of the aquifers and their characteristics. The G10M quaternary catchment remains the primary focus area, as shown in Figure 5.2.

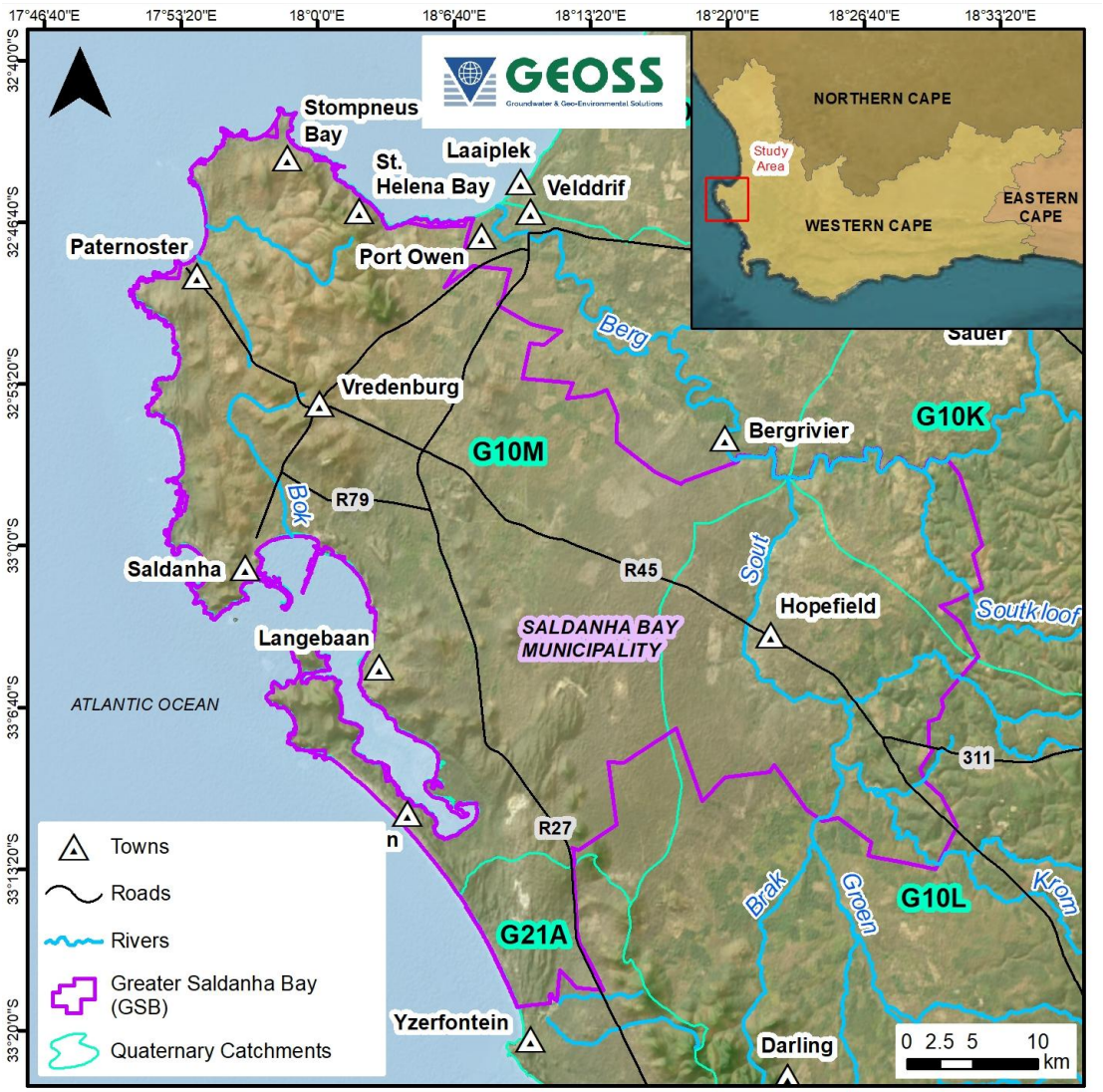


Figure 5.2. Location of the Greater Saldanha Bay study area within a regional setting.

5.2 GROUNDWATER MONITORING UPDATE

To support effective groundwater management in the area, Saldanha Bay Municipality (SBM), in partnership with GEOSS South Africa (Pty) Ltd (GEOSS), maintains a dedicated monitoring network at the municipal wellfields, complemented by an extended network of DWS monitoring boreholes at extended distances from the wellfields. The DWS borehole monitoring network was initially established with twelve (12) boreholes, and in the previous reporting period, an additional twelve (12) sites were added. These additional boreholes improve coverage around both municipal wellfields, including sites between the Langebaan Road Wellfield and the Bay, and incorporate two Geelbek boreholes. The current network layout is shown in Figure 5.5.

No changes have been made to the network over the past year, but regular monitoring continues to ensure comprehensive spatial coverage. In addition to the SBM network, the DWS also monitors groundwater at a larger network of boreholes in the region, with the locations of their sites presented in Figure 5.6. DWS monitoring data are available upon request.

5.2.1 MUNICIPAL WELLFIELDS

Saldanha Bay's water supply continues to depend largely on surface water conveyed via pipeline from dams in the Franschhoek mountain catchments. To strengthen drought resilience, two municipal wellfields, Langebaan Road (abstracting from the Lower Aquifer) and Hopefield (abstracting from the Elandsfontein Aquifer), remain in operation to supplement the bulk water supply to the Bay and surrounding towns. These wellfields also provide a critical contingency resource, ensuring that an alternative water source is available should the region experience future drought conditions.

Groundwater levels and abstraction at the wellfields are monitored monthly, with chemical parameters analysed quarterly. All production and monitoring boreholes are equipped with data loggers that record continuous water levels, which are downloaded and reported monthly.

WELLFIELD ABSTRACTION

Abstraction at both wellfields is monitored daily and reported monthly. Since their commissioning, the licensed volumes authorised by DWS have not been exceeded. The permitted annual abstraction is 2 922 000 m^3 for the Langebaan Road Wellfield and 1 602 350 m^3 for the Hopefield Wellfield. The wellfields could not be fully utilised for most of 2026 due to the vandalism that occurred in September 2025. The municipality has since worked to reestablish power at the wellfields. Many boreholes are now in working order again.

A summary graph of abstraction volumes at the Langebaan Road Wellfield between 2017 and 2026 is presented below (Figure 5.3). The highest abstraction rates were recorded in 2018 and 2019, coinciding with the drought experienced in the region. Since then, abstraction has generally decreased, though it increased slightly in 2024. To date in 2026, 8.84% of the annual licensed volume has been utilised.

Similarly, a summary graph for the Hopefield Wellfield covering 2018 to 2026 is shown in Figure 5.4. Abstraction volumes have increased steadily over the years, with the highest recorded in 2024. For 2026, approximately 2.20% of the annual licensed volume (1 602 350 m^3) has been abstracted so far.

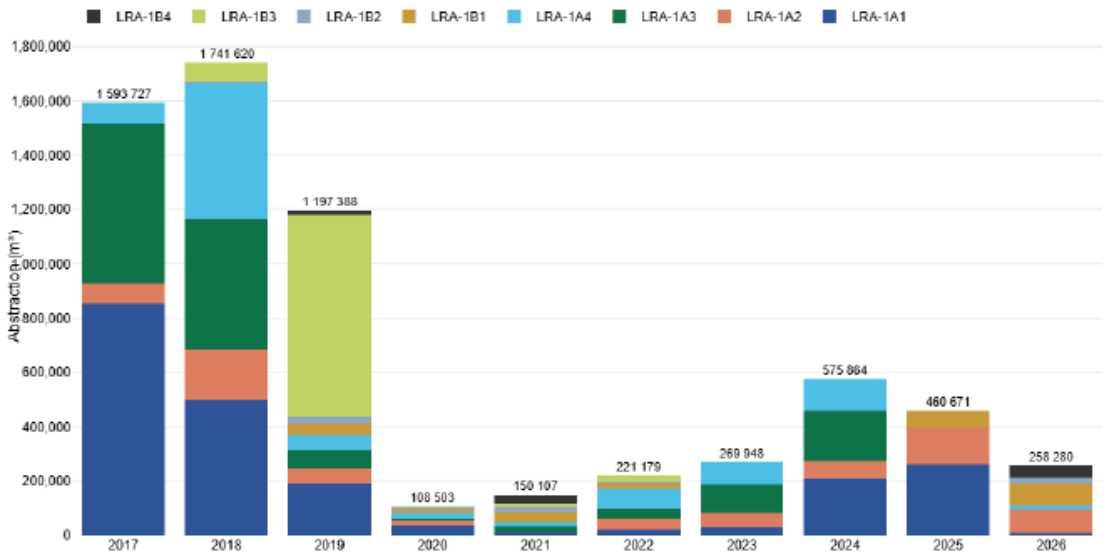


Figure 5.3. Annual abstraction from the Langebaan Road Wellfield.

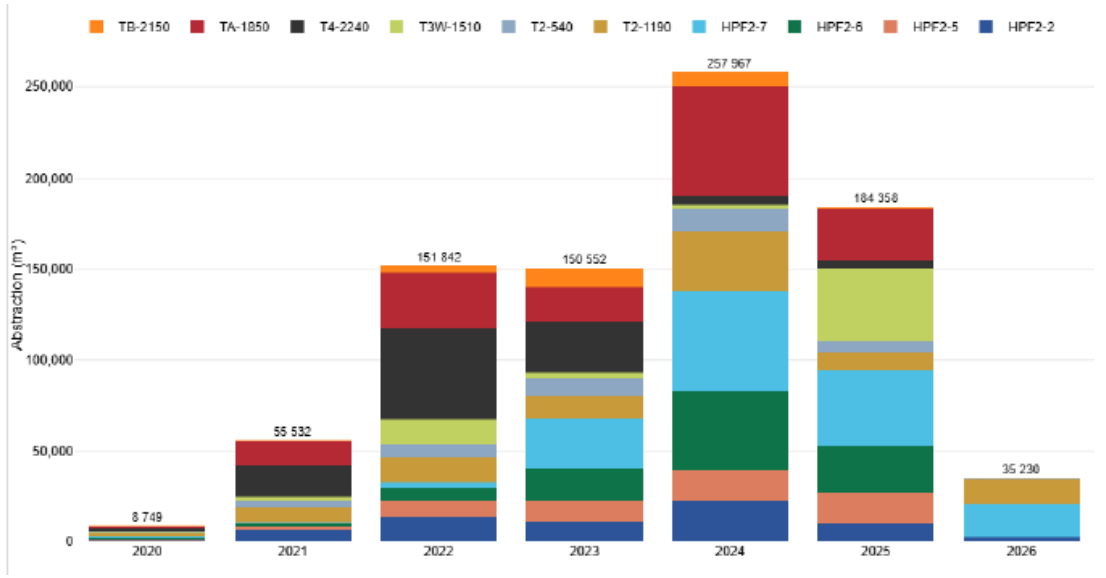


Figure 5.4. Annual abstraction from the Hopefield Wellfield.

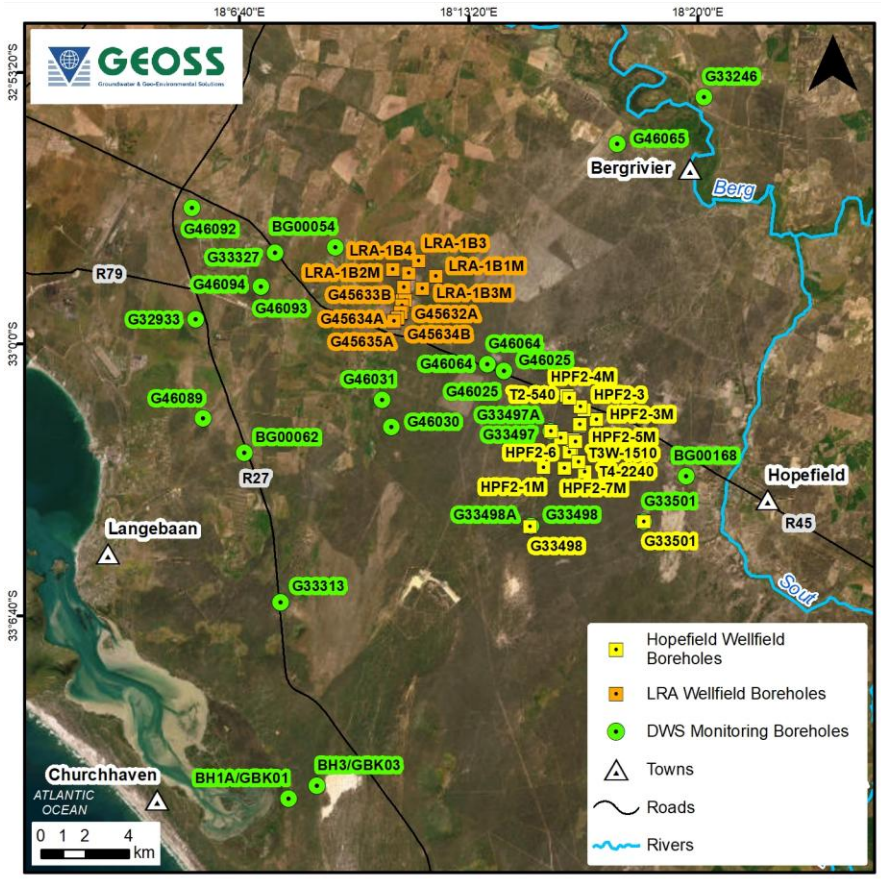


Figure 5.5. Spatial view of the locations of the selected monitoring boreholes across Saldanha Bay.

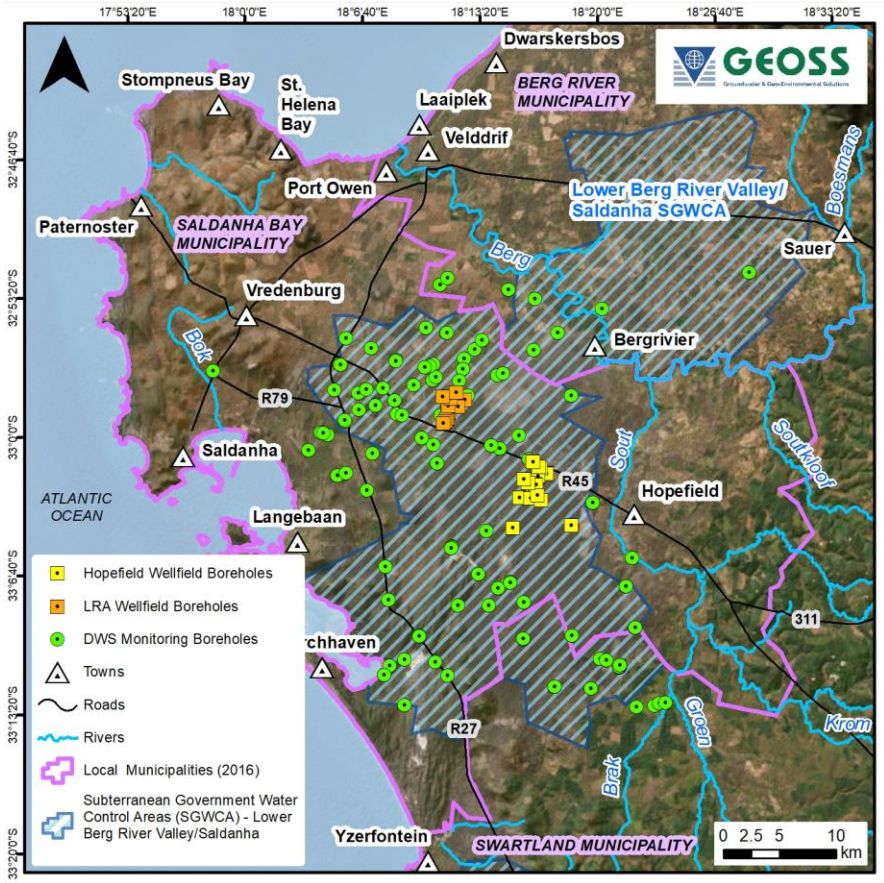


Figure 5.6. Spatial overview of the DWS monitoring boreholes within the larger area.

5.2.2 DWS/SBM Monitoring Network

The DWS monitoring borehole network, monitored monthly by SBM and GEOSS, was established at strategic locations around the wellfields. The following steps were undertaken to select boreholes for inclusion in the monitoring network:

1. Availability of data: Selected sites required long-term water level and water quality (electrical conductivity) records to evaluate groundwater trends. DWS and previous site personnel were consulted during this step.

2. Current Monitoring: DWS was consulted to identify sites currently monitored and future monitoring plans. More frequent DWS monitoring at a site would strengthen the impact assessment by ensuring more regular data collection.

3. Accessibility: Given that boreholes in the area are often distant and difficult to access, sites were selected for ease of access to ensure all boreholes could be visited efficiently during each monitoring trip.

4. Control Points: Boreholes were selected to surround key control points, ensuring that the final network adequately captures potential impacts on groundwater in Saldanha Bay.

5. Aquifer Type: To assess impacts on the various aquifers in Saldanha Bay (Upper, Lower, and Bedrock), boreholes were selected to target each aquifer specifically.

In 2024, the SBM expanded the groundwater monitoring network to address spatial gaps and improve regional coverage. While initial monitoring points were concentrated around key control areas, the expansion targeted flow paths (Figure 5.7) and enhanced spatial distribution, particularly between the Langebaan Road Wellfield and Saldanha Bay. The expanded network supports more effective monitoring of groundwater dynamics throughout the area.

The updated list of boreholes forming the monitoring network is presented in Table 5.1. These 24 boreholes will serve as the designated monitoring sites for Saldanha Bay moving forward. Groundwater levels and basic water chemistry (electrical conductivity) are measured monthly, providing an overview of the regional groundwater status.

Long-term monitoring indicates that groundwater levels across the borehole network have remained stable, with no discernible regional trends in either water levels or overall water quality, as reflected by electrical conductivity measurements. These findings suggest that current wellfield abstraction rates and other groundwater uses are not imposing significant stress on the broader aquifer systems in the region.

Localised fluctuations in water levels are evident in certain boreholes, primarily driven by seasonal recharge cycles and wellfield pumping. A small number of upper aquifer boreholes show drops in water levels linked to increased agricultural abstraction in nearby areas. In contrast, boreholes drilled into the bedrock aquifer maintain stable levels over time and show no measurable response to regional pumping activities.

Boreholes of particular importance are located at Geelbek, near the southern tip of the Langebaan Lagoon. These sites play a critical role in monitoring whether upgradient abstraction and/or mining activities could affect freshwater inflows into this part of the lagoon, which supports a unique and sensitive ecosystem. Monitoring data show no

indication that wellfield abstraction or upgradient mining activities are adversely affecting this area.

Hydrochemically, the upper aquifer exhibits higher electrical conductivity values (115–500 mS/m), reflecting greater groundwater mineralisation. The lower aquifer, which hosts the wellfield boreholes, generally shows lower conductivities (50–110 mS/m), except borehole G46092, situated outside the main groundwater flow path. The bedrock aquifer displays the widest range of conductivity (60–800 mS/m), reflecting greater variability in water-rock interactions and local recharge conditions.

Long-term groundwater level data (updated to June 2026) for the control points and analysis can be found in Section 13.13: Appendix 1.

Table 5.1. Selected monitoring boreholes for the Saldanha Bay area.

	BH ID	Latitude	Longitude	Aquifer
1	G46064	-33.00985	18.23050	Langebaan Road (lower)
2	G46031	-33.024110	18.179020	Langebaan Road (unknown)
3	G46092	-32.944730	18.087620	Langebaan Road (lower)
4	G33327	-32.963480	18.127610	Langebaan Road (unknown)
5	BG00054	-32.961460	18.157050	Langebaan Road (unknown)
6	G46065	-32.933970	18.271270	Langebaan Road (lower)
7	G46025	-33.012720	18.238470	Langebaan Road (upper)
8	G33313	-33.106630	18.128970	Elandsfontein (upper)
9	G46030	-33.035270	18.183600	Bedrock (shale)
10	BG00062	-33.045360	18.111760	Bedrock (granite)
11	G33246	-32.901000	18.336530	Adamboerskraal (upper)*
12	G33326	-32.933970	18.271270	Langebaan Road (upper)
13	G46093	-32.97735	18.12070	Langebaan Road (lower)
14	G46094	-32.97730	18.12074	Langebaan Road (lower)
15	G46089	-33.03110	18.09193	Elandsfontein (upper)
16	G32933	-32.9899	18.09195	Langebaan Road (lower)
17	G33497	-33.03743	18.26099	Elandsfontein (lower/deep)
18	G33497A	-33.03745	18.26098	Elandsfontein (upper/shallow)
19	G33498	-33.07605	18.25110	Elandsfontein (lower/deep)
20	G33498A	-33.07607	18.25110	Elandsfontein (upper/shallow)
21	BG00168	-33.05616	18.32655	Bedrock (shale)
22	G33501	-33.07463	18.30582	Elandsfontein (upper/shallow)
23	GBK01	-33.18704	18.13189	Elandsfontein (upper/shallow)
24	GBK03	-33.18171	18.14594	Elandsfontein (lower/deep)

*The Adamboerskraal Aquifer is the aquifer located north of the Berg River. This particular borehole is monitored to ensure that wellfield abstraction does not draw the saline water typical of this aquifer across the river, as this could adversely affect water quality.

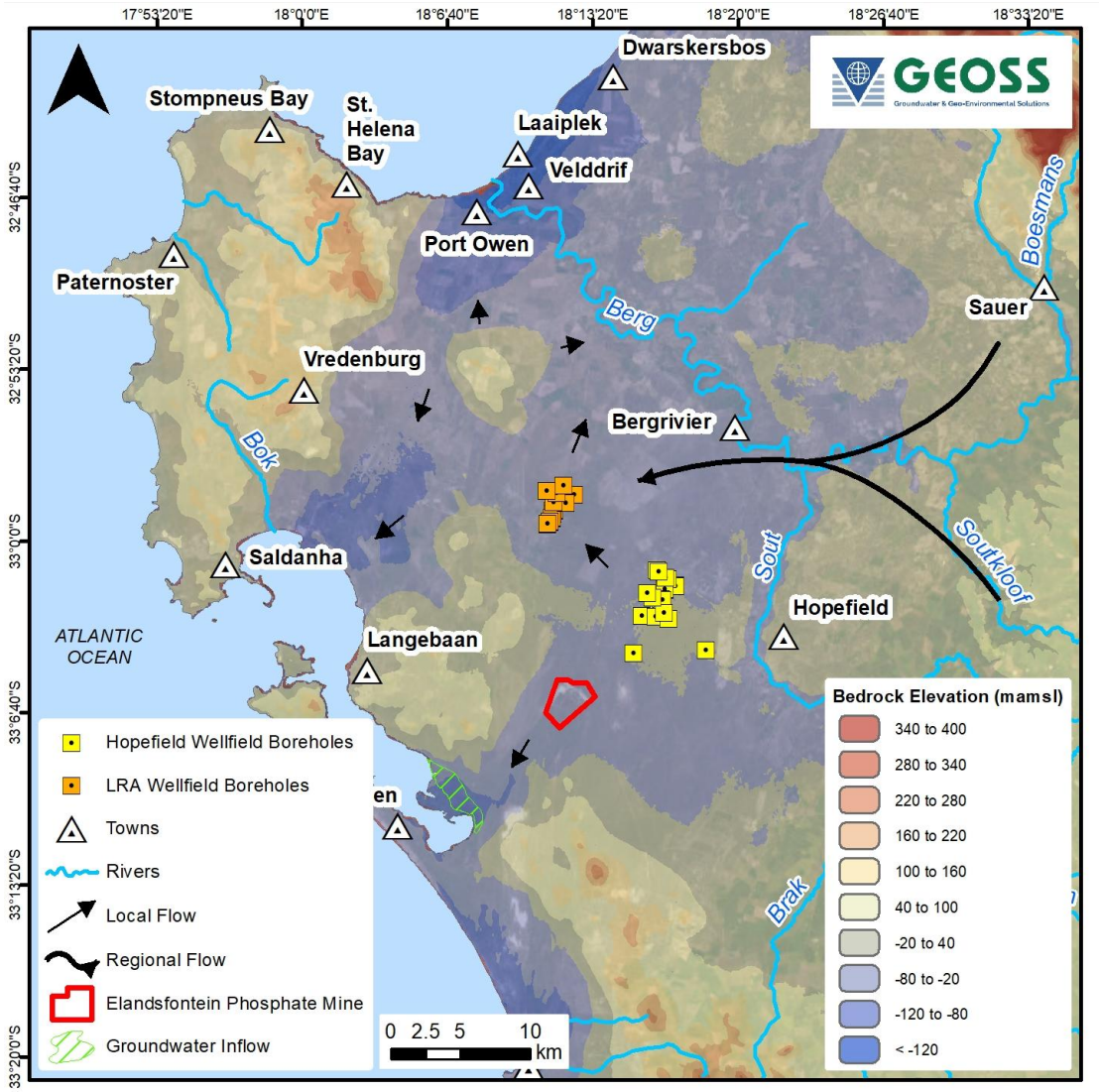


Figure 5.7. Deep groundwater flow for the lower aquifer systems in the study area.

5.3 Elandsfontein Mine and the Paleochannel to Langebaan Lagoon

Langebaan Lagoon, the sheltered arm of Saldanha Bay that lies within the West Coast National Park, has been protected as a wetland of international importance under the Ramsar Convention since 25 April 1988, in recognition of its extensive salt marshes, sand and mud flats, dwarf shrubland and islands, and its role as a major fish nursery and one of the most important sites for migratory waders and seabirds on the South African coast. The ecological character of the lagoon depends not only on tidal exchange with the open bay but on the freshwater and groundwater inputs that reach it from the surrounding catchment, so any activity with the potential to alter those inputs is directly relevant to the long-term ecological state of the bay.

A few kilometres inland of the lagoon, the Elandsfontein phosphate mine was developed over a buried paleochannel—an infilled ancient river course—whose coarse, permeable sediments form the preferred pathway along which groundwater in this part of the aquifer system moves from the interior towards Langebaan Lagoon (Figure 5.7). Regional hydrogeological studies of the wider area describe an aquifer structure of an upper and a

lower water-bearing unit separated by a clay aquitard, with the phosphate ore body sitting within the saturated zone of the upper unit. Because of this, the pit cannot be mined without actively removing (dewatering) groundwater.

Rather than discharging the pumped water, the mine operates a closed-loop dewatering-and-recharge system: water pumped from the pit is reinjected into the same aquifer via a line of injection boreholes positioned on a ridge, southwest of the mining pit, with the explicit purpose of sustaining the natural water levels and hydraulic gradients that ultimately discharge towards the lagoon (Figure 5.8).

This section uses the mine's own groundwater monitoring network to assess, in quantitative terms, how effective that system has been over the life of the mine to date. Two cross-sections along an approximately 18 km monitoring transect, running from the lagoon shoreline, across the mine, to the up-gradient (inland) limit of the monitoring network, compare pre-mining baseline conditions against a recent monitoring round, first for groundwater levels and then for electrical conductivity (EC), a general indicator of the dissolved-ion content and salinity of the water. The intent is to establish whether the influence of dewatering is confined to the pit itself, as designed, or is detectable as a trend propagating towards the Lagoon.

5.3.1 Elandsfontein Mine Monitoring Network

The transect (Figure 5.8) runs approximately 18 km from south-west to north-east, from four shallow inflow-monitoring points at the lagoon edge, inland across the R27 regional road, over a dune ridge reaching roughly 125 m above mean sea level, and into the Elandsfontein mine boundary before continuing to a set of up-gradient monitoring points to the north-east. Along this line the network includes:

- Up-gradient monitoring boreholes to the north-east of the mine, which serve as a broad indication of conditions away from both the pit and the lagoon.
- Dewatering boreholes within the active pit, which remove water to allow mining;
- The injection line, a series of boreholes positioned down-gradient of the pit that return dewatered water to the aquifer; and
- On-site and down-gradient monitoring boreholes (including the Geelbek Boreholes (GBK) and SANParks Boreholes (SNP) series and Production Boreholes (PBH)-series piezometers) between the lagoon and the mine, and immediately flanking the pit;
- Lagoon inflow monitoring points, closest to Langebaan Lagoon itself.

For each monitoring point along the transect, the mine's monitoring programme compares the pre-mining (baseline) water table and EC against current values, expressed as the change in water level in metres and the change in EC as a percentage. These are point-in-time comparisons rather than a continuous time series, and represent two snapshots of the same cross-section rather than a statistical trend.

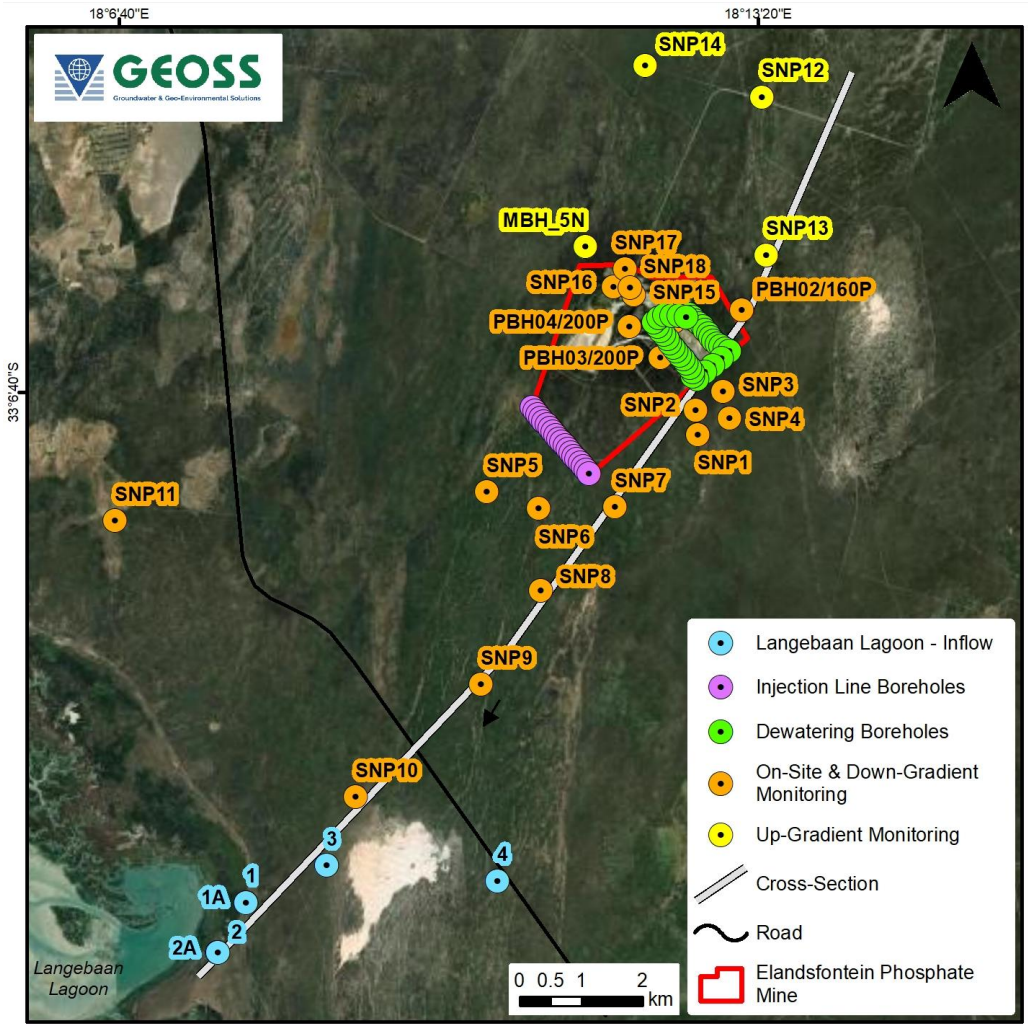


Figure 5.8. The Elandsfontein Mine groundwater monitoring network and SW–NE cross-section from the up-gradient monitoring points in the north-east, towards the Elandsfontein mine boundary (red), to the Langebaan Lagoon inflow points (blue).

5.3.2 GROUNDWATER LEVELS

Figure 5.9 presents a 2D cross-section extending from the mine towards the Langebaan Lagoon, illustrating changes in the water table between pre-mining and current conditions.

Upgradient of the mining pit (SNP12, SNP13), water levels remain virtually unchanged. Around the pit (SNP02, PBH02). The largest changes in the section occur here, as expected for an actively dewatered open pit: −7.29 m at SNP02, on the down-gradient (lagoon-facing) side of the mining area, and −6.35 m at PBH02, on the up-gradient side. This is the cone of drawdown created by pit dewatering, and both points lie within or immediately adjacent to the mine's own footprint.

Between the lagoon and the mine (SNP10, SNP09, SNP08, SNP07). Water levels here are, if anything, slightly higher than pre-mining conditions, rising from +0.32 m at SNP10 to +3.27 m at SNP08 and +3.24 m at SNP07, the point closest to the injection line. This pattern is consistent with the injection line locally sustaining the water table up-gradient of the pit, the intended effect of returning dewatered water to the aquifer in this location, rather than any adverse impact. There is no large mounding adjacent to this injection line,

which is an indication that the managed aquifer recharge is occurring successfully and that the aquifer is accepting the water.

At the lagoon edge (GBK01, GBK03), water levels are effectively unchanged from pre-mining conditions: +0.01 m at GBK01 and −0.05 m at GBK03. At the point in the system closest to the Ramsar site, there is no measurable drawdown or mounding attributable to mining.

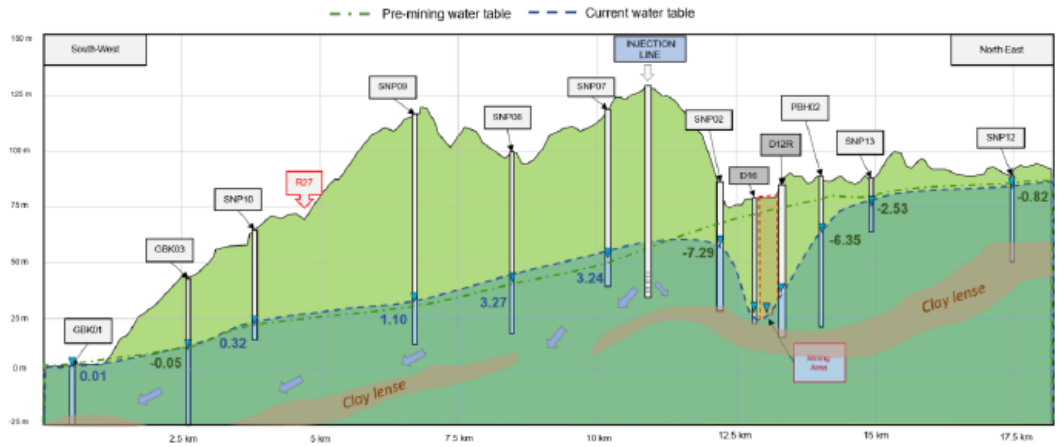


Figure 5.9. Cross-section of pre-mining and current water tables along the SW-NE monitoring transect. Labels show water-level change (m), and the red shading indicates the active mining area.

5.3.3 Electrical Conductivity

In Figure 5.10, electrical conductivity (EC) is used as a general proxy for the total dissolved-ion content, and hence salinity, of the groundwater; a rising trend in EC moving from the mine towards the lagoon would be the signature to watch for if dewatering, or the injected water, were degrading water quality along the flow path.

Up-gradient (SNP13, SNP12). Both up-gradient points show a modest decrease (improvement) in EC, of around −14.9 mS/m and −14.0 mS/m, respectively.

Between the mining pit and the lagoon (PBH02 to SNP10), changes in EC are variable: +2.7 mS/m at SNP10, +12.9 mS/m at SNP09, −2.0 mS/m at SNP08, −11.6 mS/m at SNP07, +0.7 mS/m at SNP02 and +10.1 mS/m at PBH02. None of these values is large in absolute terms, and there is no consistent, one-directional trend of increasing salinity working outward from the pit, the pattern that would be expected if reinjected or pit water were degrading aquifer water quality along the paleochannel.

At the lagoon edge (GBK01 and GBK03), decreases of 25.3 mS/m at GBK01 and 47.6 mS/m at GBK03 are noted. Both boreholes show an improvement relative to their respective monitoring records. Shallow groundwater at the lagoon edge is naturally more susceptible to seasonal recharge, rainfall variability, and tidal or estuarine influences on the near-shore water table, which may contribute to EC variability independent of inland mining activities. The observed decreases in EC at GBK01 and GBK03 are therefore considered to be within the range of natural variability and do not, in isolation, indicate a mining-related impact extending to the lagoon. Furthermore, water levels in this portion of the aquifer remain largely unchanged, while the concurrent improvement in EC is

considered positive. Overall, the current conditions at the lagoon edge do not indicate cause for concern.

Figure 5.11 shows the spatial distribution of pre-mining EC values between the Elandsfontein Mine and Langebaan Lagoon, while Figure 5.12 displays the corresponding change in EC (mS/m) between pre-mining and current conditions along the same transect.

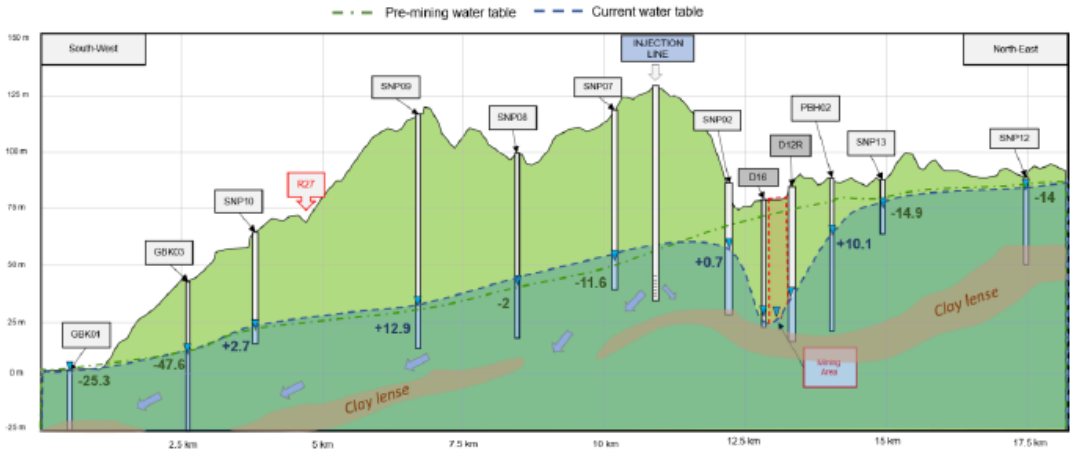


Figure 5.10. Cross-section along the same transect, showing the percentage change in electrical conductivity (EC) at each monitoring point, current relative to pre-mining.

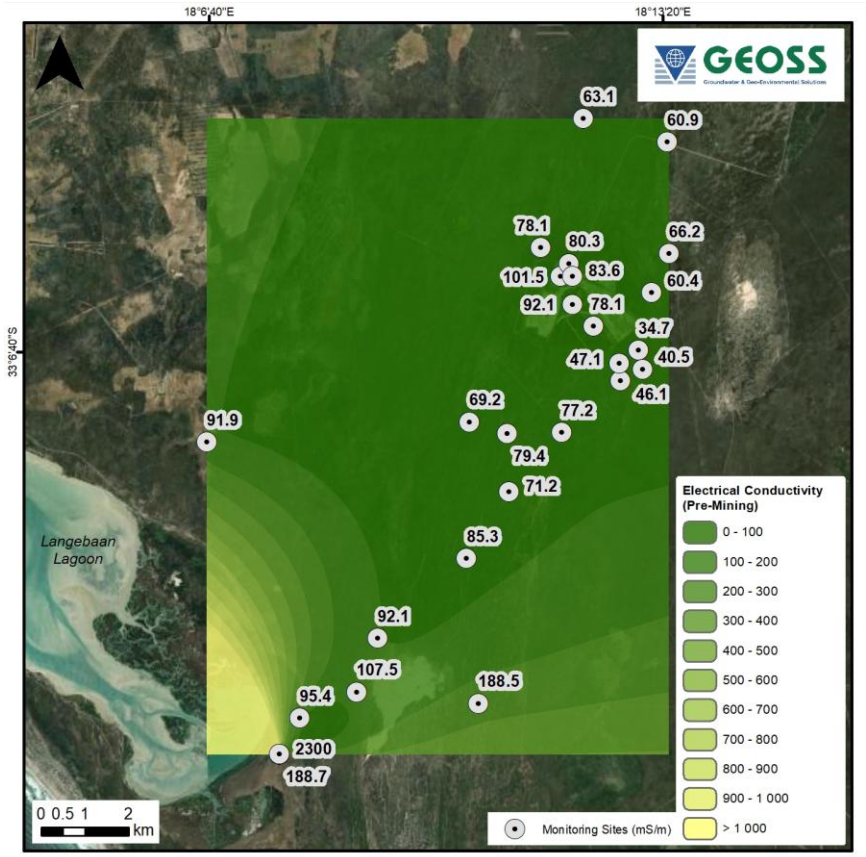


Figure 5.11. Pre-mining EC distribution between the Elandsfontein Mine and the Langebaan Lagoon.

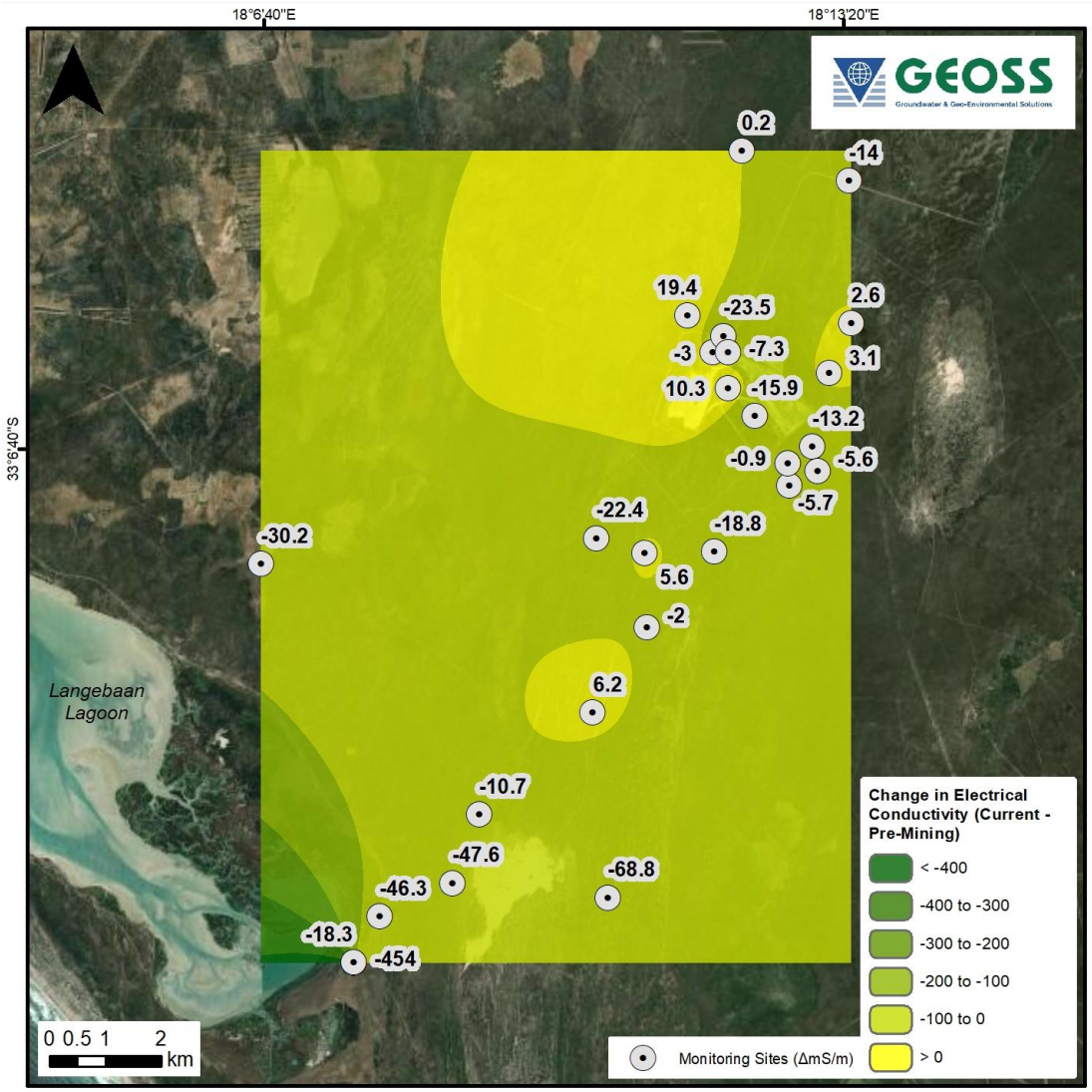


Figure 5.12. Difference in EC (mS/m) between pre-mining and current conditions along the transect from Elandsfontein Mine to Langebaan Lagoon.

5.3.4 Discussion

Read together, the water-level and EC cross-sections tell a consistent story. Groundwater levels show a real, well-defined and expected cone of drawdown centred on the actively dewatered pit (−6 to −7 m at the two boreholes adjacent to the mining area), which recovers to within a metre of pre-mining conditions by the time the transect reaches its up-gradient limit some 6 km further north-east, and shows no discernible drawdown at all at the lagoon end of the section. If anything, water levels between the lagoon and the mine are slightly elevated relative to pre-mining conditions, consistent with the injection line performing its intended function of returning water to the aquifer up-gradient of the pit.

Electrical conductivity shows a broadly similar picture of stability: minor changes across the transect, with no consistent trend linking the mine to the lagoon. The larger EC changes recorded at the two lagoon-edge boreholes are, on the balance of the evidence in this dataset, most plausibly attributable to natural variability at that location rather

than to mining, given both their direction (a decrease, i.e. freshening) and the absence of any intervening trend that would connect them to mining activities.

Collectively, this is consistent with the intended function of the closed-loop dewatering-and-injection system at Elandsfontein: groundwater removed from the pit to permit mining is returned to the same aquifer down-gradient, sustaining the hydraulic gradient along the paleochannel that ultimately discharges towards Langebaan Lagoon. The monitoring data reviewed here do not show the pattern that would indicate the system is failing to perform this function.

Table 5.2 summarises the change in water level and EC at each of the ten labelled monitoring points along the transect.

Table 5.2. Change in groundwater level (m) and electrical conductivity (mS/m) relative to pre-mining conditions at monitoring points along the SW–NE cross-section. Negative values indicate decreases from pre-mining conditions.

Monitoring point	Approx. distance along section	Position relative to the mine	Δ Water level (m)	Δ EC (mS/m)
GBK01	~0.3 km	At the lagoon shoreline monitoring line	+0.01	-25.3
GBK03	~2.7 km	Near the lagoon, south-west of the mine	-0.05	-47.6
SNP10	~3.5 km	Between the lagoon and the mine	+0.32	+2.7
SNP09	~6.7 km	Between the lagoon and the mine	+1.10	+12.9
SNP08	~8.5 km	Approaching the injection line	+3.27	-2.0
SNP07	~10.3 km	Adjacent to the injection line	+3.24	-11.6
SNP02	~12.2 km	Immediately south-west (down-gradient side) of the pit	-7.29	+0.7
PBH02	~14.7 km	Immediately north-east (up-gradient side) of the pit	-6.35	+10.1
SNP13	~15.7 km	North-east of the mine, up-gradient	-2.53	-14.9
SNP12	~18.3 km	Far north-east, up-gradient limit of the section	-0.82	-14.0

5.4 Conclusion

To support effective groundwater management in the area, the SBM, in partnership with GEOSS, maintains a dedicated monitoring network at the municipal wellfields, complemented by an extended network of DWS monitoring boreholes located further afield. The network has remained unchanged over the past year, but regular monitoring ensures continued comprehensive spatial coverage.

Abstraction at both wellfields is monitored daily and reported monthly. Since commissioning, the licensed volumes authorised by DWS have not been exceeded. The permitted annual abstraction volumes are 2 922 000 m^3 for the Langebaan Road Wellfield and 1 602 350 m^3 for the Hopefield Wellfield. By mid-2026, approximately 8.84% of the Langebaan Road Wellfield's licensed volume and 2.20% of the Hopefield Wellfield's licensed volume have been abstracted, demonstrating continued compliance and responsible abstraction. The low abstraction volumes to date are largely due to extensive vandalism at both wellfields in September 2025.

Hydrochemically, the upper aquifer exhibits higher electrical conductivity values, reflecting greater groundwater mineralisation. The lower aquifer, which hosts the wellfield boreholes, generally shows lower conductivities, except for borehole G46092, located outside the main groundwater flow path. The bedrock aquifer displays the widest conductivity range, reflecting greater variability in water-rock interactions and local recharge conditions.

Long-term monitoring indicates that groundwater levels across the borehole network remain stable, with no discernible regional trends in water levels or overall water quality, as reflected by electrical conductivity measurements. This demonstrates that current abstraction and other groundwater uses are not imposing significant stress on the broader aquifer systems. Boreholes of particular importance are located at Geelbek, near the southern tip of the Langebaan Lagoon. These sites are critical for detecting any potential impacts of upgradient abstraction or mining on freshwater inflows to this ecologically sensitive area. Monitoring data show no evidence of any adverse effects, indicating that the lagoon's unique ecosystem is not currently under threat from groundwater use or nearby mining activities.

This conclusion is reinforced by the cross-section of monitoring boreholes running from the Elandsfontein mine to the Langebaan Lagoon (Geelbek). Since dewatering and managed aquifer recharge began at the mine, groundwater levels and electrical conductivity along this paleochannel pathway to the Lagoon have remained close to pre-mining conditions at the lagoon end, with change concentrated near the pit itself. This is a reassuring result for a Ramsar-listed system whose ecological character depends in part on stable groundwater inputs, and is consistent with the mine's closed-loop dewatering-and-injection system performing as designed. It should nonetheless be read as a current snapshot rather than a permanent guarantee: continued, regular monitoring of this transect, together with the wider borehole network, remains the most direct way of confirming that the Lagoon continues to be protected as mining and abstraction continue.

The local monitoring database is updated monthly, while the regional database is updated quarterly.

Deploying a CTD in Saldanha Bay

6 WATER QUALITY

6.1 INTRODUCTION

Coastal water quality is a fundamental indicator of marine ecosystem health, directly affecting biodiversity, ecosystem services, human health, and economic activities such as fisheries, mariculture, and tourism. Globally, increasing anthropogenic pressures e.g., urbanisation, port expansion, industrial discharges, and climate change are leading to measurable changes in coastal water quality, particularly in closed semi-enclosed embayments or more sheltered environments (Islam & Tanaka 2004). In such coastal systems, these pressures are compounded by alterations to natural hydrodynamic regimes, often resulting in reduced flushing, organic enrichment, and hypoxic or anoxic events (Gray et al. 2002).

Continued, regular water quality monitoring is essential when assessing the health of marine ecosystems. Changes in water quality can have implications for both marine flora and fauna, and for human health. Monitoring of key water quality parameters can help detect pollution sources and emerging contaminants early, allowing for timely intervention and remediation interventions (Chapman & Sullivan 2022). Temperature, salinity, and dissolved oxygen (DO) concentration in marine waters are the variables most frequently measured water quality parameters which help us understand the physical and biological processes impacting on, or occurring within, a body of seawater. Within Saldanha Bay and Langebaan Lagoon regular monitoring of water quality in these systems has been conducted for several decades to track changes and inform management responses. Historical records dating back to the 1970s provide valuable insight into long-term trends in these key physico-chemical variables, as well as the prevalence of microbial and trace metal contamination.

Port expansions, including the construction of the Marcus Island causeway and the Iron Ore Terminal (IOT), have altered the natural hydrodynamic regime of Saldanha Bay, influencing wave exposure, circulation patterns, and retention times, particularly in Small Bay. These changes, combined with effluent discharges, organic loading from mariculture, and increased stormwater runoff, have the potential to degrade water quality. The impacts of these pressures are further compounded by regional oceanographic dynamics, such as seasonal upwelling and broader climate trends, including increasing sea temperatures and wave energy. In this study, we assess recent water quality data (up to July 2026) from Saldanha Bay and Langebaan Lagoon. We focus on parameters that are routinely monitored and for which long-term datasets exist, including temperature, salinity, DO, microbial contamination (*E. coli* and faecal coliforms), and concentrations of trace metals in bivalve tissues. These data are contextualised within broader historical datasets to evaluate long-term environmental change and inform management priorities.

Here, we synthesise the most recent evidence on the current state of water quality in the Bay and Lagoon, evaluate compliance with regulatory thresholds relevant to public health and mariculture, and identify emerging trends that may require further investigation or management intervention. The findings are critical for informing the sustainable management of coastal resources in the region, ensuring long-term health of

these important ecosystems and underpinning several chapters in the State of the Bay (SOB) programme.

6.2 CIRCULATION, CURRENTS AND CLIMATE

The currents and circulation of Saldanha Bay have previously been described by Weeks et al. (1991a). The authors 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. The authors also concluded that the harbour construction has 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 IOT (Figure 6.1).

It was also revealed that greater wind velocities do indeed influence current strength and direction throughout the water column (Weeks et al. 1991b). Strong NNE winds were observed to enhance the surface flowing SSW currents along the IOT 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.

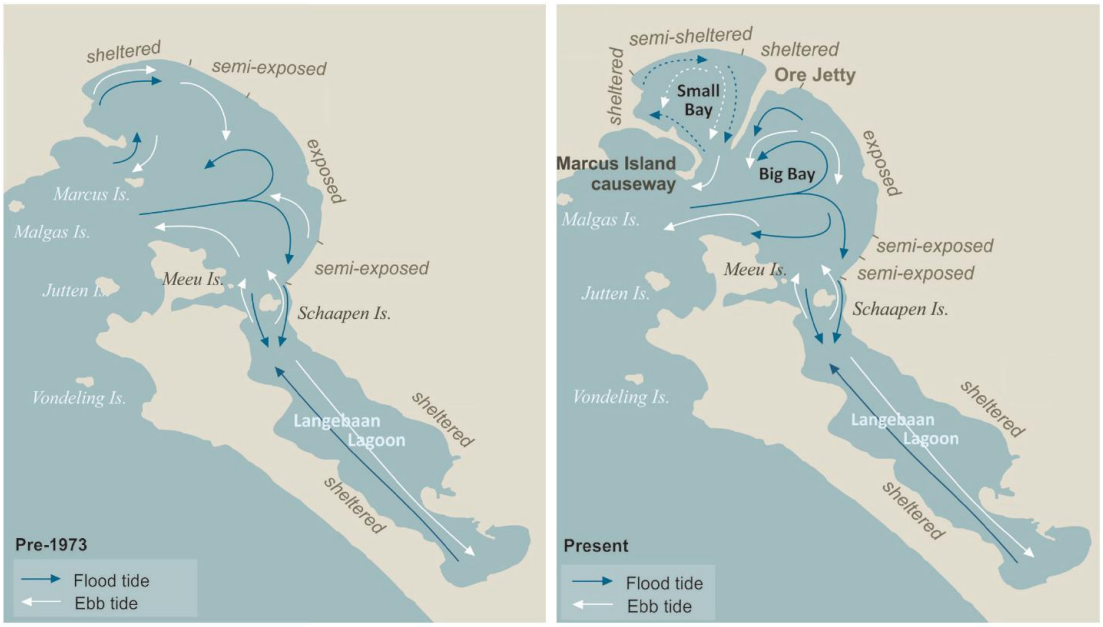


Figure 6.1. Representation of the surface currents and circulation of Saldanha Bay prior to harbour development (pre-1973, left), and after construction of the Marcus Island causeway and Iron Ore Terminal (present, right) (Adapted from: Shannon & Stander (1977) and Weeks et al. (1991a)).

Construction of the IOT and the causeway to Marcus Island had a major impact on the distribution of wave energy in Saldanha Bay, particularly in Small Bay. Prior to port development, four wave-energy zones were distinguished within the Bay, defined as 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 (Flemming 1977) (Figure 6.2 left). The IOT essentially divided the Bay into two, 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. Wave exposure in Big Bay was altered less dramatically, however, the extent of sheltered and semi-sheltered areas increased after harbour development (Luger et al. 1999). This is pertinent when interpreting water quality results and identifying pollution processes and inputs around the Bay.

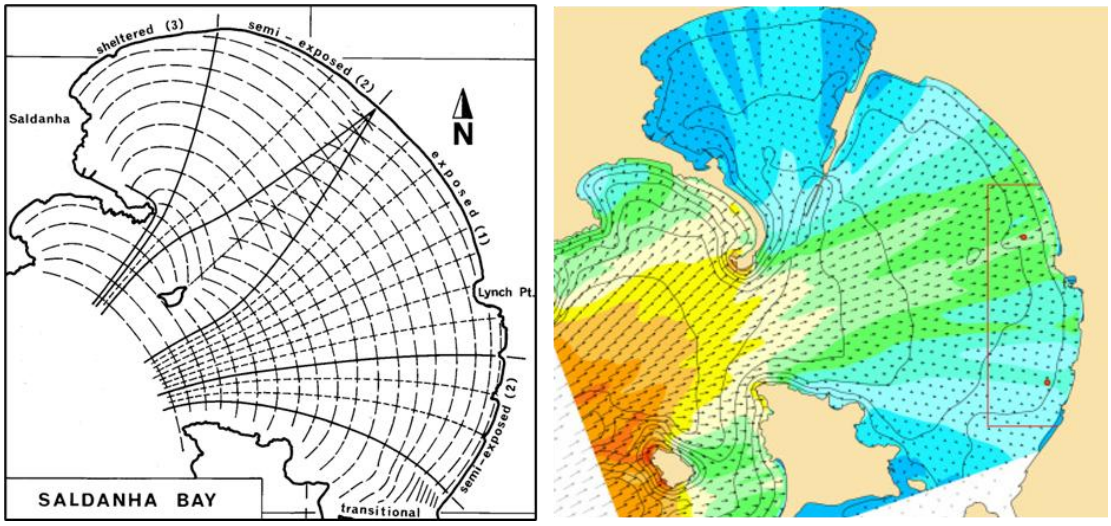


Figure 6.2. Predicted wave fields in Saldanha Bay showing wave height and direction prior to (left) and post (right) harbour development. Orange shading indicates wave heights >1.4 m, while blue shading indicates wave heights of <0.6 m (Sources: Flemming (1977) and WSP (2010)).

Regarding the current wave climate, daily mean wave direction of ocean/sea surface waves data were accessed from two sites close to Saldanha Bay, for Jan 1959–Dec 2021 (ERA5 data from Hersbach et al. (2023)) as was significant wave height (combined wind and swell) data which was also coupled with measured 'Significant wave height' (SWH) data from a WaveRider buoy moored within Saldanha Bay.

Mean wave direction at these sites, including the frequency occurrence of each wave height class (0–9 m), is expressed in Figure 6.3. It is evident that over ≈ 60 years, dominant wave direction is approximately SW, with wave heights largely between 0–3 m (mean height = 2.34 m).

SWH is the average height of the highest third of surface ocean/sea waves generated by wind and swell (Hersbach et al. 2023). Daily SWH data for the two locations were averaged and data were then aggregated by decade (latest data = 2022). The frequency distribution (Weibull probability density function (PDF)) of SWHs for each decade is presented (Figure 6.4 left). The Weibull PDF is a statistical distribution used to model the distribution of continuous random variables (Weibull 1951). The data show a marginal shift over time towards higher SWHs occurring more frequently, from the 1960s–2022, with the 1980s showing particularly greater frequency of higher SWHs (Figure 6.4 left). Maximum significant wave height observed within each decade also shows a general increase over time, with the highest observed SWH occurring on 13 July 2020 (Table 6.1).

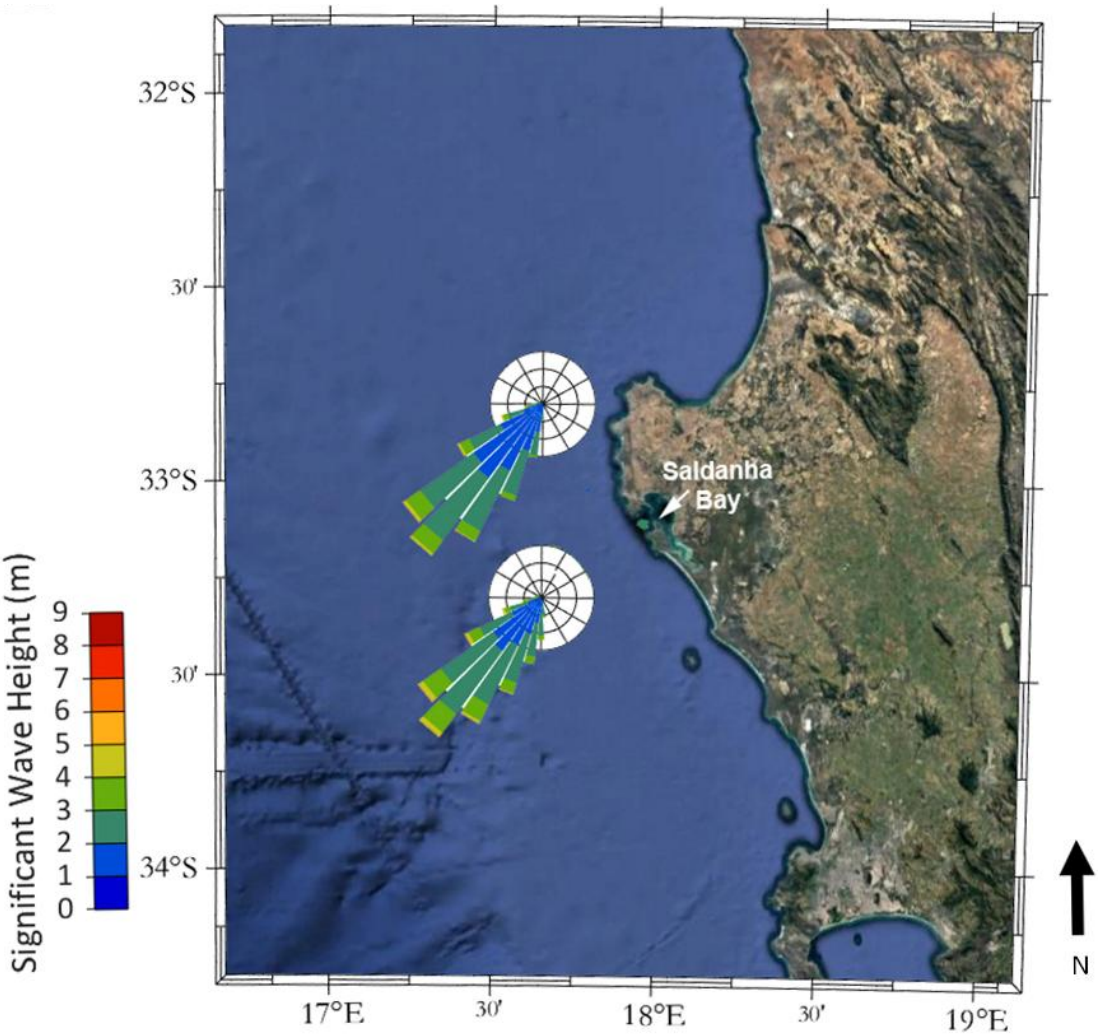


Figure 6.3. Compass roses (average direction x frequency of observed wave height) for daily ERA5 data (12:00 PM each day) January 1959–December 2021, for two locations nearest to Saldanha Bay (Hersbach et al. 2023).

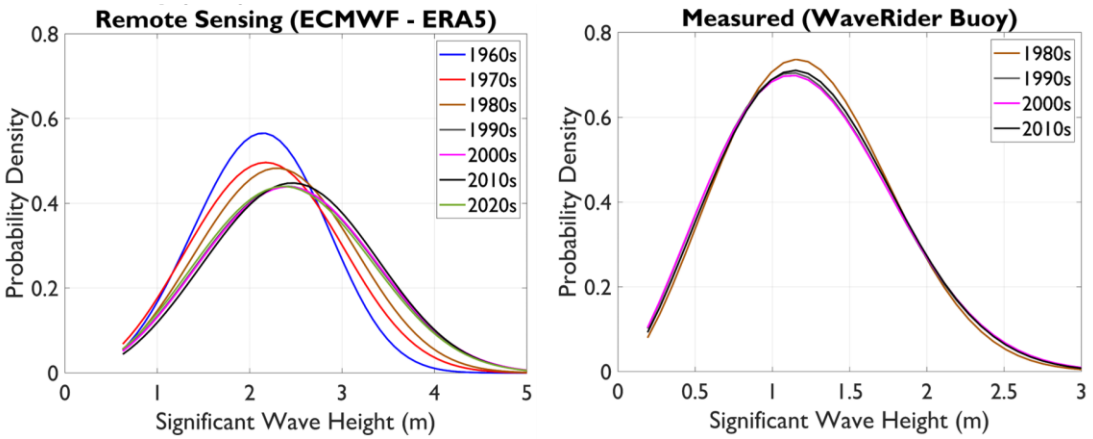


Figure 6.4. Left: Frequency distribution (Weibull PDF) of significant wave height (SWH) in meters for aggregated locations close to Saldanha Bay for each decade (daily data from 1959–2022). Right: Data sourced from the CSIR Non-Directional WaveRider buoy deployed in Saldanha Bay (33.05°S, 17.978°E).

Satellite derived measurements such as these are effective sources of information, however, linking this to in situ ground truthing further enhances interpretation. Wave climate variability was further investigated through the analysis of wave height data gathered by a non-directional WaveRider buoy that is permanently moored near the entrance channel to Saldanha Bay, near the Marcus Island causeway just outside the shipping channel. The WaveRider data series available for analysis spans from 1981–2020. As the WaveRider buoy provides non-directional data only SWH outputs were compared. As with the remotely sensed data, maximum SWH observed within each decade showed a general increase over time (Figure 6.4 right).

Extreme wave height as measured by the WaveRider buoy is the sum of the highest peak and deepest trough and gives an indication of the most extreme wave instance within a decade. Extreme wave height also showed a general increase over time, with two rogue waves occurring in the late 1990s (Table 6.1).

Table 6.1. Maximum Significant Wave Height (SWH) observed for each data set within each available decade.

Decade	Maximum SWH: ECMWF-ERA5	Maximum SWH & (Extreme Wave Height): CSIR WaveRider	Average SWH: ECMWF-ERA5
1960s	4.18		1.61 (n=3 654)
1970s	5.02		1.66 (n=3 651)
1980s	5.96	5.26 (9.34)	1.74 (n=3 654)
1990s	5.43	5.36 (15.54)	1.83 (n=3 652)
2000s	5.85	6.15 (9.99)	1.84 (n=3 653)
2010s	6.10	8.03 (11.55)	1.85 (n=3 652)
2020–2022	6.24		1.82 (n=731)

Both datasets show a shift over time towards higher SWHs occurring more frequently in recent decades. Remotely sensed data indicate that the greatest shift in wave climate occurred between the 1960–1980s. Unfortunately, the WaveRider dataset does not begin until 1981 and therefore there is no way to validate this finding without additional in-situ data.

These data support published literature that the wave climate around South Africa is becoming more extreme, and that this, coupled with increasing sea level and more extreme temperatures, is indicative of a shifting global climate (Mather 2012, DEA 2014, Muis et al. 2016, Allison et al. 2022).

6.3 WATER TEMPERATURE

Monitoring of water temperature within Saldanha Bay has been undertaken by various organisations as part of a range of surveys conducted over the past 50 years. Detailed descriptions of these surveys can be found in previous SOB reports (Clark et al. 2023) and the published literature (Monteiro & Brundrit 1990, Monteiro et al. 1999).

In summary, the temperature of surface waters in Saldanha Bay fluctuates seasonally, with surface sun warming occurring in summer, followed by cooling in winter, while the temperature of deeper (>10 m) water shows smaller magnitude, non-seasonal, variation,

with previously recorded summer and winter temperatures being quite similar (Shannon & Stander 1977, Monteiro & Brundrit 1990, Luger et al. 1999). In most years, a strong thermocline (an abrupt temperature gradient in a body of water, marked by a layer, above and below which the water is at different temperatures) separating the sun warmed surface layer from the cooler deeper water is typically present during the summer months between 5–10 m depth. During the winter months, this thermocline often breaks down due to surface cooling and increased turbulent mixing, and the water column becomes nearly isothermal (surface and deeper water similar in temperature) (Monteiro & Brundrit 1990, Monteiro & Largier 1999). Furthermore, the summer thermocline is not a long-lasting feature, but has a six to eight day cycle (Monteiro et al. 1997). 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 the 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.

Warm oceanic water is typically more saline and nutrient-deficient than the cool upwelled water that usually occurs below the thermocline. This is reflected in the high salinity and low nitrate and chlorophyll concentrations (a measure of phytoplankton production) previously observed (Weeks et al. 1991a). The influx of cold, nutrient rich, upwelled water into Saldanha Bay is therefore critical in sustaining primary productivity within the Bay, with implications fishing and mariculture activities. 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) influx from the Benguela upwelling system. As a result, these shallow water areas do not support large plankton blooms and are usually visually clear.

With a view to maintaining a long-term temperature dataset within Saldanha Bay, Anchor deployed five underwater temperature recorders, programmed to record temperature every hour, at 2 m, 4.5 m, 7 m, 9.5 m and 12 m depths (a 'thermistor array'). The array is retrieved and serviced annually (since 2014). Average daily temperature data for the period August 2018–March 2026 are presented in Figure 6.5. It should be noted that during the array's deployment, it has periodically suffered technical failures or data losses (including in 2025/2026 where, for example, the 4.5m and 7m loggers are not complete datasets). Therefore, during various short periods of time, some data from certain depths are missing (Figure 6.5). To address some these gaps as far as possible, data from a bottom water temperature logger located near Small Bay control site (<100 m away from array) were provided by the Department of Forestry, Fisheries and the Environment (DFFE) (comparisons showed temperatures were fairly consistent between the two loggers, despite marginally different depths (14 m v 12.5 m) and locations in Small Bay (Clark et al. 2022).

The temperature data presented a similar pattern to the 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 (Figure 6.5). 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. The data also shows a clear seasonal trend, with water temperature being much higher in summer (Dec–Feb, maximum 28°C) than in winter (Jun–Aug, minimum 10°C) (Figure 6.5). Although not presented here, diurnal fluctuations in temperature are similar across all seasons, with temperatures increasing over the course of the day, peaking in the early afternoon, then declining through the afternoon and night, reaching a minimum at the time of sunrise each day.

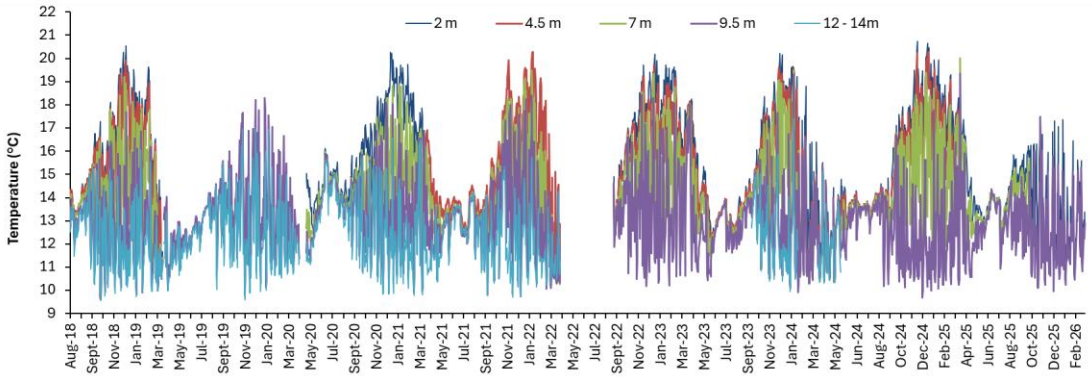


Figure 6.5. North buoy (Small Bay) temperature time series for the period August 2018–March 2026. Temperatures were recorded every hour, and the average daily temperature is shown here (includes data from Anchor thermistor array and DFFE loggers).

There appears to be no clear trend of seawater warming or cooling over time, but rather some anomalies creeping into the normal seasonal scale events , which may be linked with El Niño–La Niña climatic cycles. Continuous water temperature monitoring is valuable for analysing these long-term trends. This is an economically viable way of detecting changes in the frequency of anomalous conditions, such warm oceanic water intrusion events, which would have significant impacts on productivity, health and provision of ecosystem services around the Bay.

6.4 SALINITY

6.4.1 SALDANHA BAY

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 & Nelson 1996), and the salinity values recorded for the Bay usually fall within this range. During summer months when wind driven coastal upwelling 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 Saldanha Bay cover much of the same period as that for water temperature. These studies showed there is 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. 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). This highlights the impacts of fluctuations in physical oceanography (water temperature and salinity) from the immediate area on local biological processes (nitrate and chlorophyll) occurring within Saldanha Bay (Luger et al. 1999).

6.4.2 LANGEBAAN LAGOON

Given the sensitivity of the Langebaan Lagoon, and in spite of confidence expressed by a range of specialists that groundwater use by Elandsfontein mine is unlikely to impact on the Lagoon (see Chapter 5: Groundwater), Kropz Elandsfontein in conjunction with the WQT have been monitoring a range of biological and physico-chemical variables associated with Langebaan Lagoon to establish an appropriate baseline against which any potential future changes in the Lagoon can be benchmarked. This includes monitoring of temperature and salinity (see below) and biota (see Chapter 8: Benthic Systems) as well as macrophytes (see Chapter 12: Aquatic Macrophytes) around the top end of the Lagoon (Clark et al. 2022).

An Aqua TROLL 200 data logger was installed at the end of August 2018 and has been collecting near continuous temperature and salinity data. Average annual data from 2019 to 2026 are presented on Figure 6.6 below, along with measured water level (note some of the 2020 data was not available as the instrument was stolen partway through that year).

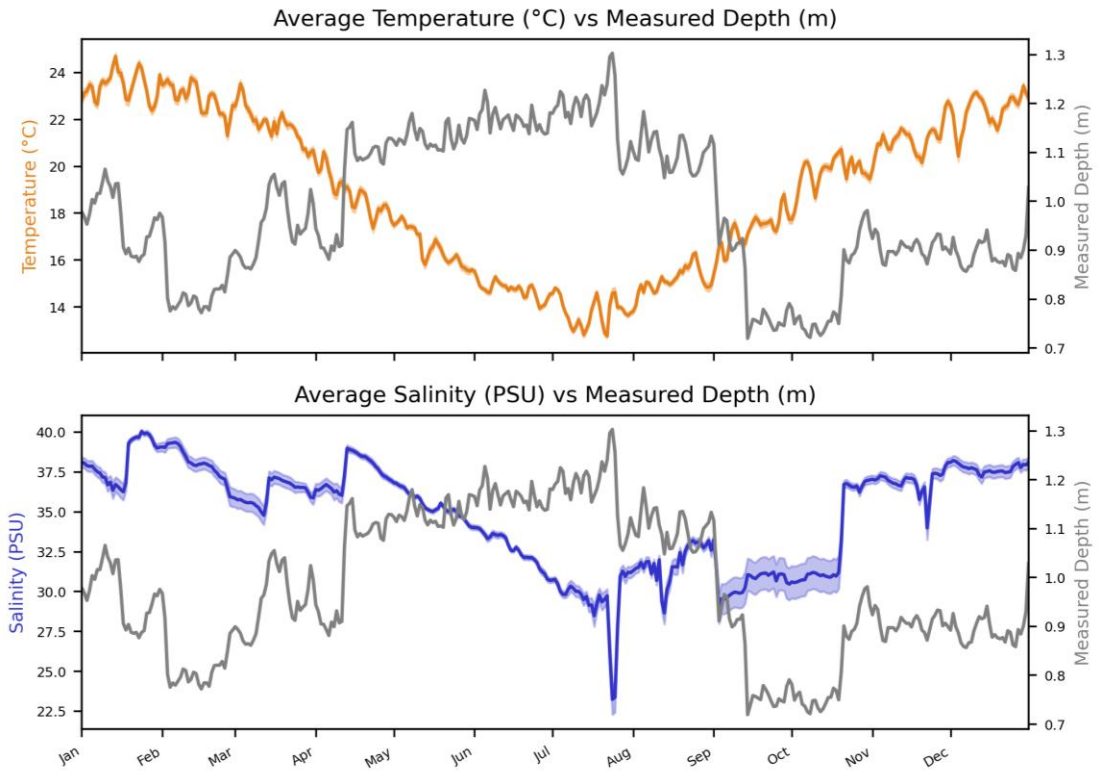


Figure 6.6. Variations in temperature (°C, top, orange) and salinity (PSU, bottom, blue) (Annual average 2019–2026). Data taken from Anchor's Aqua TROLL 200 moored at the head of Langebaan Lagoon.

The data shows periods (summer months) of hypersaline conditions (Figure 6.6). Water emerging from the marshes at the head of the Lagoon becomes severely hypersaline as a result of evaporation in summer, and even though it may be diluted by freshwater flowing out of the aquifer, this is not sufficient to bring the level below that of normal seawater. It is likely that this effect (development of hypersaline conditions) is quite localised at present (i.e., restricted to the extreme upper reaches of the Lagoon only) but could become much more pervasive if freshwater outflow from the aquifer were to drop in future.

The trend in salinity exhibits a similar diurnal oscillation to that for temperature, but this oscillation is linked to the state of the tide (rather than the time of day) and changes through the year. In winter, salinity oscillates between that of normal seawater (around 35.0 PSU) at high tide and a slightly fresher state (between 32.0 and 34.0 PSU) at low tide. Salinity appears to drop as the tide recedes. This suggests that fresh water from the aquifer is flowing into the lagoon as the hydraulic gradient allows this with the dropping tide. In summer, the pattern reverses with salinity increasing from that of normal seawater (35.0 PSU) at the peak of the high tide, becoming hyper-saline (39–40 PSU) as the tide recedes. It is likely that this is a function of increased evaporation at this time of year (linked to higher prevailing air temperatures) in conjunction with the lower volume of water at low tide.

6.5 Dissolved Oxygen

Sufficient dissolved oxygen (DO) in seawater is essential for the survival of nearly all marine organisms. Low oxygen (or near anoxic conditions (<0.5 ml/l) 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 (Lamberth et al. 2010). Once all the oxygen in the water is depleted, anaerobic bacteria (not requiring oxygen) continue the decay process, causing the characteristic sulphurous smell. These extreme “black tide” events are not known to occur in Saldanha Bay, but anthropogenic activities have resulted in low oxygen conditions at times.

Small Bay experiences fairly regular oxygen deficit events during the summer months, as does Big Bay, but the events in Big Bay are typically less frequent and of lower magnitude (Weeks et al. 1991a). Weeks et al. (1991a) attributed the oxygen deficit in Small Bay largely to anthropogenic causes, namely reduced flushing rates (due to the Marcus Island causeway and IOT construction) and discharges of organic rich effluents. Oxygen levels within Small Bay are very low during the late summer to autumn months, likely as a result of naturally occurring conditions i.e., increase in oxygen deficit during summer and follow sustained intrusions of cold bottom water at which time the bay is intensely stratified and bottom water renewal is reduced (Pitcher et al.). 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 mariculture mussel rafts and within the yacht basin), caused by excessive organic inputs. For detailed descriptions of historical trends observed in previous surveys of DO levels within Saldanha Bay, please see the 2021 SOB report.

The most recent DO measurements (2023–2026) are from ongoing deployments of RINKO W DO sensors 0.5 m off the seafloor at mariculture farm and control sites in both Small Bay and Big Bay, by DFFE (Figure 6.7). These data show that these frequently hypoxic conditions (<2 ml/l) continue later through the year, until autumn (late May/early June) which, as described above, relates to the decline in upwelling and increased turbulent mixing of the water column with the onset of winter leading to higher DO levels in the bottom waters. It is also clear that there are regular hypoxic and even anoxic events recorded during summer and autumn in both Bays. A major increase in the frequency of Small Bay hypoxic events occurred after the harbour development in the 1970s which reduced flushing rates, and the situation hasn't changed much since (Clark et al. 2020).

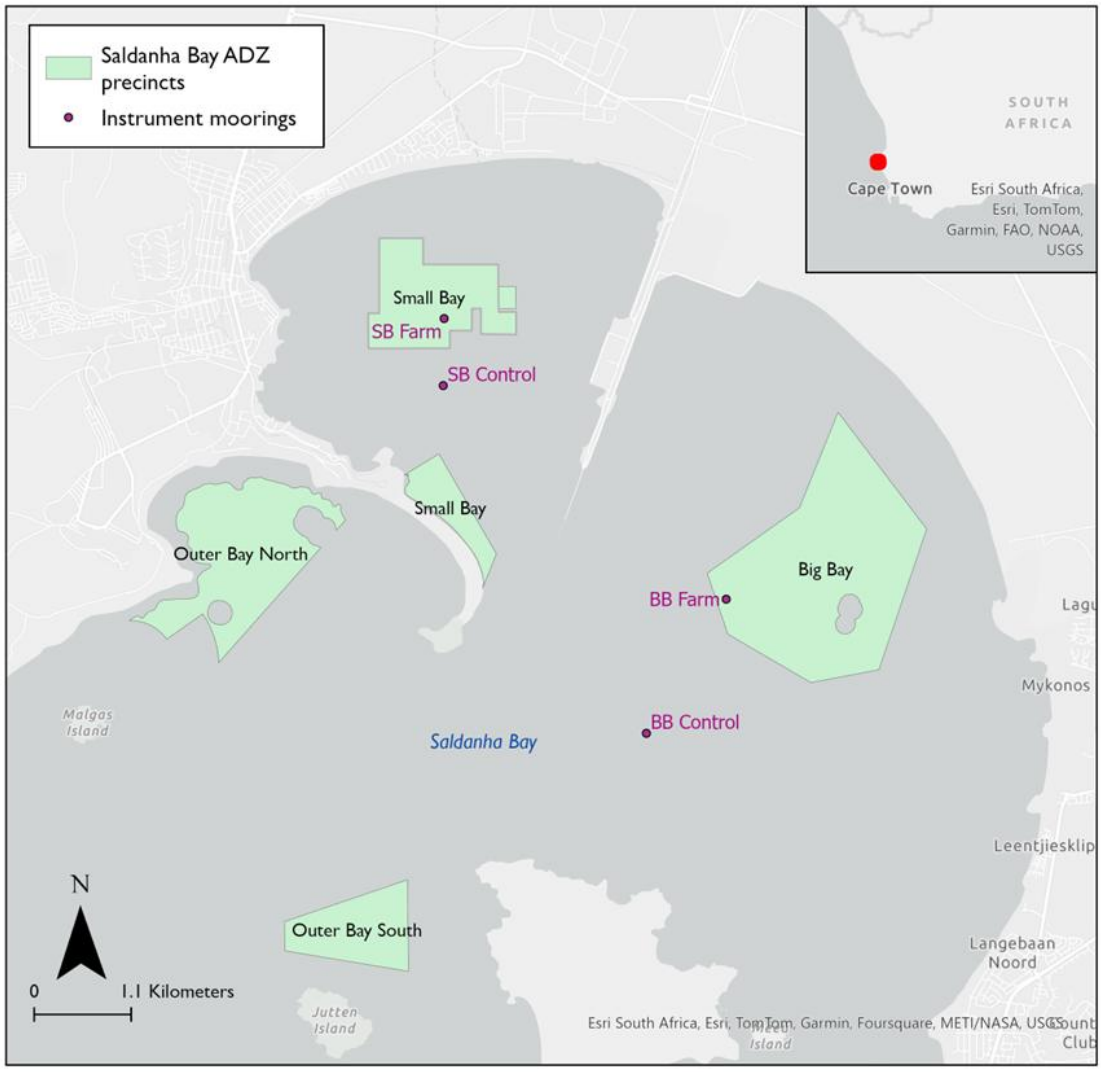


Figure 6.7. Saldanha ADZ precincts (labelled in black text) and location of moored oxygen sensors (labelled in maroon).

There is high spatial variability in near bottom DO concentration throughout Saldanha Bay, largely due to variation in hydrodynamic process and anthropogenic organic inputs. The recent data support historic observations that indicate that Big Bay experiences regular hypoxic events during autumn, reflecting the external source of low oxygen upwelled water (Figure 6.8). The absence of any severe anoxic events, however, is

attributed to better mixing of the water column in Big Bay, lower retention times (enhanced flushing) and lower organic loading from anthropogenic sources (e.g., mariculture, fish factory effluent, wastewater treatment works). Note that currently part of the Big Bay ADZ precinct has Environmental Authorisation for finfish cage farming, yet no farming has commenced (partly on account of the sensitive reef habitat directly below the fin fish area, see Section 8.2.2, Chapter 8: Benthic Systems). In other areas of the ADZ of the precinct operators have Marine Rights for both oysters and mussels, which are currently being grown on longlines and rafts.

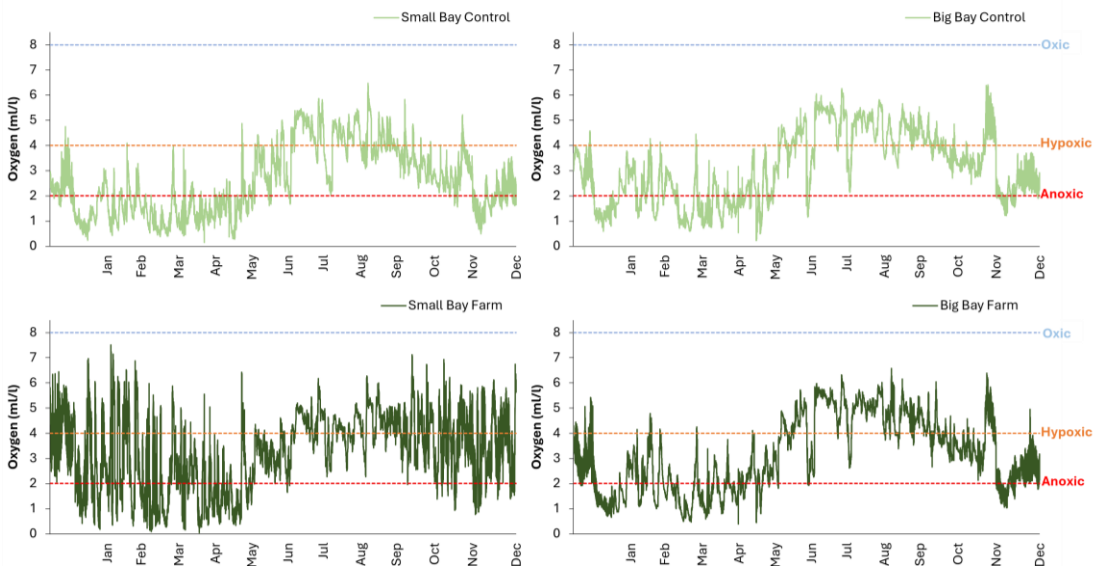


Figure 6.8. Comparison of Dissolved Oxygen concentrations at mariculture Control (top) and Farm (bottom) sites in Small Bay (left) and Big Bay (right), from Jan 2023–July 2026 (15-minute average). Data courtesy of DFFE (Department: Spatial Assessment and Management).

Within Small Bay, oxygen concentrations at the Farm site (8 m depth) were typically lower than the Control site (14 m depth), despite being substantially shallower (DO typically decreases with depth so the deeper control site is expected to have lower DO levels under natural conditions) (Figure 6.8). The more pronounced and frequent hypoxia at the Small Bay Farm sites is considered to be a result of organic loading and higher microbial respiration, i.e., a consequence of bivalve mariculture. Hypoxic and intermittent near anoxic conditions in the lower part of the water column are frequent occurrences during summer-autumn seasons in Small Bay and anthropogenic organic loading appears to exacerbate the situation. Near-anoxic conditions were observed on the Small Bay Farm site, while Control site DO concentrations, although also hypoxic, remained higher and did not display a complete depletion of oxygen (0 ml/l) (Figure 6.8). Conditions at the Small Bay Farm site are more variable in general, especially throughout summer, but even through periods of hypoxia (Figure 6.8). It should be noted that the instrument moored at the Small Bay farm is shallower (8 m) than the Small Bay control site (12 m) so despite the potentially influence of organic loading, this site is more variable and is more influenced by regular ventilation of bottom waters which likely dilutes some of the impact on DO from the farm related nutrient input. This also explains the regular hypoxic conditions of the control site which is less well flushed which increases the chances of an oxygen deficit build up (Pitcher et al. in press). The higher incidence of severely hypoxic and in

some cases anoxic bottom waters, accompanied by more corrosive low pH conditions and altered nitrate reduction pathways as shown by (Pitcher et al. in press)., within the farmed areas of Small Bay demonstrates the need to exercise caution in increasing shellfish production in Small Bay (Pitcher et al. in press).

6.6 Biotoxins and Harmful Algal Blooms

Saldanha Bay is naturally susceptible to harmful algal blooms (HABs), which are a recurrent feature of the southern Benguela Upwelling System (Pitcher & Calder 2000). The semi-enclosed nature of the Bay, together with seasonal coastal upwelling, high nutrient availability, relatively long water residence times and periods of water column stratification, provides favourable conditions for phytoplankton blooms to develop. Although HABs can occur throughout the year, they are most common during the austral spring through to autumn (Pitcher & Calder 2000). While many blooms are benign, some are dominated by toxin-producing phytoplankton, particularly *Alexandrium catenella*, which produces paralytic shellfish toxins (PSTs). These toxins accumulate in filter-feeding shellfish, including mussels and oysters, posing a significant risk to human health through paralytic shellfish poisoning (Grattan et al. 2016, Neves et al. 2021). Importantly, HABs are a natural feature of the highly productive Benguela ecosystem and should not necessarily be interpreted as evidence of anthropogenic pollution or deteriorating water quality, although nutrient enrichment from human activities may influence the intensity or persistence of some bloom events.

The DFFE undertakes routine phytoplankton and shellfish toxin monitoring throughout Saldanha Bay in recognition of the importance of the region's aquaculture industry. This monitoring provides an early warning of toxin-producing blooms and ensures that shellfish harvesting is suspended when toxin concentrations exceed regulatory safety limits. In May 2026, monitoring detected a significant bloom of *A. catenella*, with PST concentrations in farmed mussels and oysters exceeding the regulatory limit by more than fifteen-fold[5]. This resulted in the temporary closure of shellfish harvesting areas in Saldanha Bay and a public warning against the collection or consumption of wild shellfish from the area.

While PST-producing blooms represent the principal risk to shellfish aquaculture, other HABs may also affect marine ecosystems by causing oxygen depletion or gill irritation, occasionally resulting in fish mortalities. Consequently, HABs can have ecological, public health and economic consequences through impacts on fisheries, aquaculture production and recreational harvesting. As HABs are a natural and recurring feature of the region, continued phytoplankton surveillance and shellfish toxin monitoring remain essential components of environmental management, public health protection and the sustainable operation of Saldanha Bay's internationally important mariculture sector.

[5] https://www.news24.com/southafrica/news/steer-clear-of-west-coast-shellfish-due-to-high-levels-of-toxin-dept-warns-20260529-0705

6.7 MICROBIAL INDICATORS

6.7.1 BACKGROUND

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. Faecal coliforms and *Escherichia coli* are used to detect the presence of faecal pollution, but they provide indirect evidence of the possible presence of water borne pathogens and may not accurately represent the actual risk to water users (Monteiro & Largier 1999, Carr & Rickwood 2008). Using different indicators for these two receptors is scientifically justified because they represent different exposure routes and are based on different regulatory frameworks (Table 6.2).

Different microbiological indicators were therefore applied according to the intended water use: *E. coli* was used to assess recreational water quality, while faecal coliforms were used to assess suitability for mariculture, consistent with South African guideline frameworks that recognise the differing exposure pathways and management objectives for these receptors (Table 6.2).

Table 6.2. Microbial indicators used in this assessment for different water uses.

Water use	Primary microbiological indicator	Rationale
Marine recreational (bathing) waters	*E. coli*	Indicators of human health risk from accidental ingestion during swimming.
Mariculture/shellfish growing waters	Faecal coliforms	Used to assess faecal contamination because shellfish concentrate bacteria and viruses from the surrounding water. This aligns with international shellfish sanitation practices. (Department of Water and Sanitation)

6.7.2 WATER QUALITY GUIDELINES

Human pathogens are microbiological organisms which may cause disease or other health problems in humans (mainly gastrointestinal problems). In terms of marine waters, this can either be through contact or ingestion of water containing these organisms (recreation), or through the consumption of seafood, which has been cultured in contaminated waters (mariculture). For example, filter feeding organisms, such as shellfish, can accumulate pathogenic organisms in their bodies and thereby infect the people that consume them.

Marine water quality is assessed according to the most sensitive water use applicable to the specific area (e.g., mariculture vs. industrial use). The South African Water Quality Guidelines For Coastal Marine Waters–Volume 1: Natural Environment and Mariculture Use (DFFE 2022) provides updated recommended guidelines for faecal coliforms in water bodies used for mariculture.

The Recreational WQGs (DEA 2012) 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 (DEA 2012). 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. Analyses should be completed by an accredited laboratory, preferably one with ISO17025 accreditation.

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. 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 6.3.

Table 6.3. 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/100 ml)
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)

The South African Water Quality Guidelines For Coastal Marine Waters–Volume 1: Natural Environment and Mariculture Use (DFFE 2022) guidelines aim to protect consumers of shellfish from bacterial contamination. These regulations specifically adopt a site-specific, "reference system" approach, defining 'ambient condition' "*as the natural range of conditions at a particular site (seasonal and/or event driven)*" (DFFE 2022). In the 2022 guidelines, for mariculture, i) faecal coliform concentrations shall not exceed a median value of 14 cfu (Membrane Filter), ii) with not more than 10% of the samples exceeding 31 cfu (90th percentile to be calculated), iii) nor exceed 800 cfu on any one day.

Compliance with mariculture guidelines were assessed by comparing faecal coliform counts to the DFFE 2022 guidelines (DFFE 2022), whilst recreational use compliance was assessed by comparing *E. coli* count data to the revised recreational guidelines (DEA 2012).

6.7.3 Historical Monitoring

In 1998 the CSIR were contracted by the former SBWQFT to undertake fortnightly sampling of microbiological indicators at 15 stations within Saldanha Bay. The initial

report by the CSIR, indicated that there was indeed a health risk associated with the collection and consumption of filter-feeding shellfish (mussels) in Small Bay (with fewer counts in Big Bay, with the exception of Paradise Beach).

Since 1998, regular monitoring of microbiological indicators within Saldanha Bay has continued to the present day and is now undertaken by the WCDM. The available data cover the period February 1999 to July 2026 for 20 stations (ten in Small Bay, five in Big Bay and five in Langebaan Lagoon) but an additional four sites, inside of the Lagoon, were added in 2026 (however, these sites were only sampled from June 2026 onwards so there are not enough data to report on in this year's SOB report). Data during 1998/1999–2026 period have, for the most part (at the original 20 sites), been collected on a monthly or bimonthly basis. In previous SOB reports, data were presented cover a complete calendar year to account for seasonal differences. The 2019–2026 reports, however, include data up until end May/June which therefore includes both summer and winter data. For the full data series please refer to previous editions of the SOB report (Clark et al. 2025). Here, we present microbial monitoring data for the past 10 years only.

6.7.4 2026 Water quality results–recreational use

Recreational water quality rankings for all sampled sites throughout Saldanha Bay and Langebaan Lagoon for the entire sampling period are shown in Table 6.4.

Data from the microbial monitoring programme has previously shown that generally nearshore coastal waters in the system have improved considerably for recreational use since 2005 (Table 6.4, Clark et al. (2022)). In terms of *E. coli* count data 2026 represents the best year to date, with 19 of 20 sites rated to have 'Excellent' water quality (Table 6.4). This shows an improvement on last year. Encouragingly some of the bathing beaches that were shown to have dropped in water quality last year have returned to Excellent levels, including Langebaan Main Beach, Leentjiesklip and Tooth Rock (Table 2.4). The Bok River site has consistently had the highest *E.coli* counts, and this remains the case in 2026 samples. Notably, Hoedjiesbaai, previously a notable site of Poor water quality in terms of *E.coli* maintained its Excellent water quality rating for the second year in a row. Reuse of the majority of treated wastewater from the Langebaan and Wastewater Treatment Works for other uses (including industrial, construction and irrigation) that was historically discharged into the Langebaan MPA was, until recently, having a positive effect.

The ratings provided in Table 6.4 are based on percentiles and so counts at these sites can have instantaneous reported values as high as >1500 cfu/100 ml, suggesting actual counts could be much higher for these sites at times which could be related to once-off or isolated events. This value far exceeds the threshold for water safe for recreational activity. In the event of results indicating large exceedance of guideline limits immediate action should be taken to ensure an improvement is achieved. Encouragingly, only Bok River had counts of this magnitude (two occasions in September 2025 and December 2025) since the last SOB report.

Table 6.4. Sampling site compliance for recreational use based on *E. coli* counts for sites in Small Bay, Big Bay and in Langebaan Lagoon (2010–2026). 'ND' indicates that there were insufficient data for the calculation of Hazen percentiles in that year and 'Ex.' indicates excellent water quality.

Site	1. Beach at Mussel Rafts	2. Small Craft Harbour	3. Sea Harvest - Small Quay	4. Saldanha Yacht Club	5. Pepper Bay - Big Quay	6. Pepper Bay - Small Quay	7. Hoedjies Bay Hotel - Beach	8. Beach at Caravan Park	9. Bok River Mouth - Beach	10. General Cargo Quay - TNPA	11. Seafarm - TNPA	12. Mykonos - Paradise Beach	13. Mykonos - Harbour	14. Leentjiesklip	15. Langebaan North - Leentjiesklip	16. Langebaan - Main Beach	17. Langebaan Yacht Club	18. Tooth Rock	19. Kraalbaai North	20. Kraalbaai South
1999	Fair	Ex.	Fair	Poor	Poor	Poor	Fair	Fair	Poor	Ex.	Ex.	Ex.	Fair	ND	Ex.	ND	ND	ND	ND	ND
2000	Fair	Fair	Fair	Poor	Fair	Fair	Fair	Fair	Fair	Fair	Fair	Fair	Fair	ND	Fair	ND	ND	ND	ND	ND
2001	Ex.	Good	Ex.	Poor	Poor	Fair	Poor	Fair	Poor	Ex.	Ex.	Ex.	Ex.	Good	Good	Fair	ND	ND	ND	ND
2002	Ex.	Ex.	Ex.	Fair	Fair	Good	Fair	Poor	Poor	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	ND	ND	ND	ND
2003	Ex.	Ex.	Fair	Poor	Fair	Ex.	Good	Ex.	Poor	Ex.	ND	Ex.	Fair	Good	Ex.	Ex.	ND	ND	ND	ND
2004	Ex.	Ex.	Ex.	Poor	Fair	Good	Poor	Fair	Poor	Ex.	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Fair	ND	ND
2005	Ex.	Good	Fair	Poor	Fair	Ex.	Poor	Poor	Poor	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	ND
2006	Ex.	Ex.	Ex.	Ex.	Poor	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	ND
2007	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Fair	Good	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	ND
2008	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Poor	Ex.	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	ND
2009	Ex.	Ex.	Good	Ex.	Fair	Good	Fair	Fair	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Fair	ND	ND
2010	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Fair	Fair	Good	Ex.	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	ND
2011	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Poor	Fair	Ex.	Ex.	ND	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	ND	ND
2012	Ex.	Good	Ex.	Ex.	Good	Good	Poor	Poor	Poor	Ex.	Ex.	Ex.	Ex.	Fair	Poor	Ex.	Ex.	Ex.	Ex.	Ex.
2013	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Fair	Good	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Good	Ex.	Ex.	Ex.
2014	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Good	Fair	Good	Ex.	Ex.	Ex.	Good	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
2015	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Good	Fair	Ex.	Ex.	Ex.	Ex.
2016	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Fair	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Ex.
2017	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Fair	Fair	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Good	Ex.	Ex.	Ex.
2018	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Fair	Fair	Ex.	Ex.	Ex.	Ex.	ND	Ex.	ND	ND	ND	ND	ND
2019	Ex.	Ex.	Ex.	Ex.	Ex.	ND	Poor	Fair	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.
2020	Fair	Ex.	Ex.	Ex.	Poor	Ex.	Poor	Fair	Good	Ex.	Ex.	Fair	Ex.	Ex.	Fair	Good	Ex.	Ex.	Ex.	Fair
2021	ND	Fair	Ex.	Fair	Good	Ex.	Poor	Ex.	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
2022	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Good	Poor	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
2023	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Fair	Poor	Fair	Ex.	Good	Fair	Fair	Fair	Fair	Fair	Good	Ex.	Ex.
2024	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Ex.	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
2025	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Poor	Ex.	Fair	Ex.	Ex.
2026	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.

6.7.5 2026 WATER QUALITY RESULTS–MARICULTURE USE

Data from the microbial monitoring programme have previously shown that 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 (outdated) DWAF 80th percentile mariculture guideline limits for faecal coliforms. 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) that met standards every year over the period 2000–2025. The Saldanha Bay Yacht Club has also shown considerable improvement and has met guidelines in 10 out of the past 11 years.

Recent (2016–2026) mariculture faecal coliform concentrations for sites sampled in both Big Bay and Small Bay, where mariculture is currently undertaken, are shown in Figure 6.9 to Figure 6.11.

In Small Bay, faecal coliform concentrations in 2026 were generally quite poor, with nine out of 10 sites exceeding the 10% limit (i.e., more than 10% of samples has concentrations >31 cfu/100ml (Figure 6.9, Figure 6.10). Four sites (Hoedjiesbaai, Pepper

Bay, Beach at Caravan Park and Bok River Mouth) had the worst water quality exceeding 2 or all 3 of the mariculture limits (Figure 6.9, Figure 6.10). Excluding Bok River Mouth, counts from the past year of testing did not, however, exceed the 800 cfu limit, suggesting there are no persistent local contaminant sources within the vicinity of these three sites (Figure 6.10). The more stringent guidelines suggest that water quality at Pepper Bay is worse than previously assessed, with a higher level of non-compliance over the past 10 years. Water quality at this site does, however, fluctuate considerably between years (Figure 6.9).

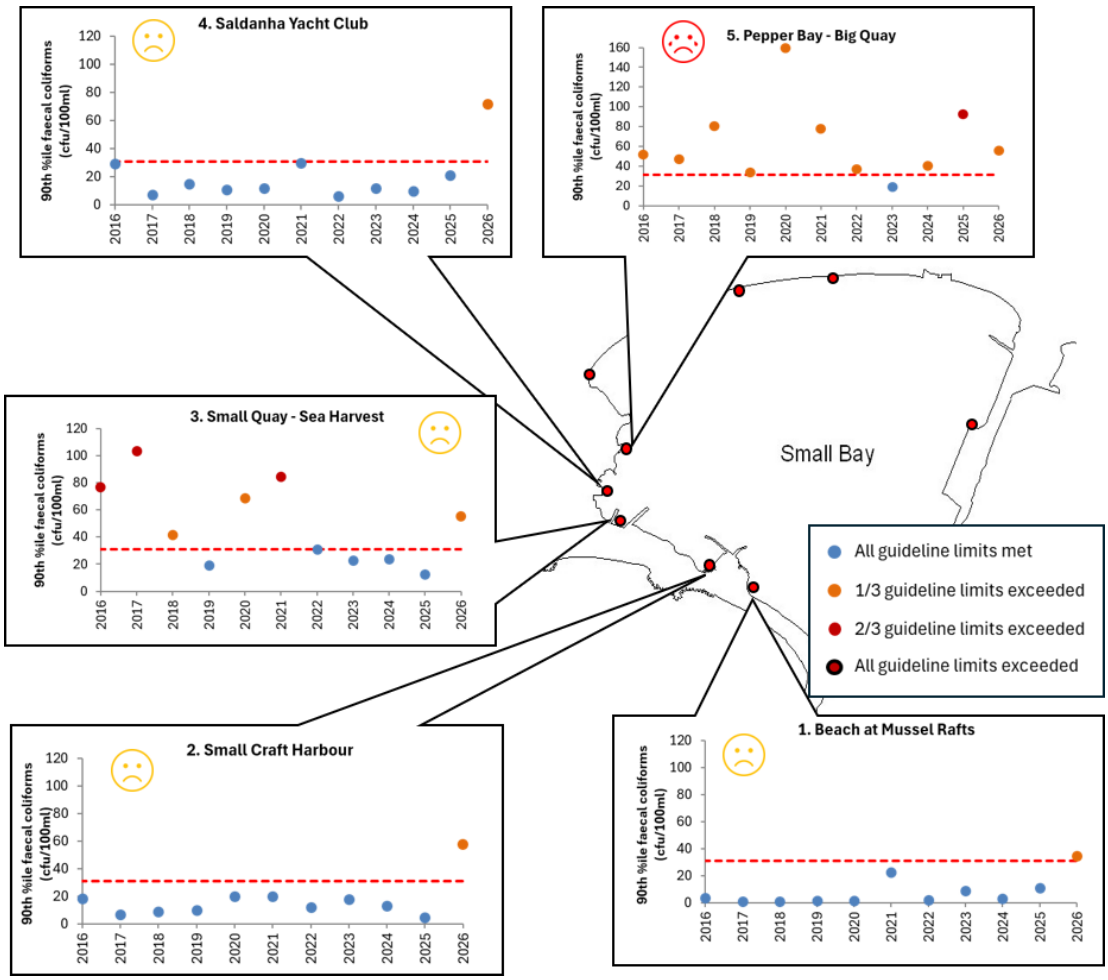


Figure 6.9. Annual 90th percentile faecal coliform counts at five of the 10 sampling stations within Small Bay (2016–2026). Red line = cfu/100 ml guideline limit (DFFE 2022), below which 90% of the count data should fall. Guideline 2: Annual counts median <14 cfu; Guideline 3: not exceed 800 cfu on any one day. All guideline limits met = blue dot, one guideline exceeded = orange dot, two guidelines exceeded = red dot, all limits exceeded = **red** dot black outline. The faces correspond to current water quality status taking into account current and historical levels.

For Hoedjiesbaai and Bok River Mouth sites, water quality remains poor, with concentrations of faecal coliforms in 2026 for Hoedjiesbaai in particularly being quite high (90th percentile: Hoedjiesbaai = 157 cfu; Bok River Mouth = 531 cfu). The areas of particular concern have always been at Hoedjiesbaai and the Bok River Mouth (despite recreational contact limits being met and water rated as Excellent for Hoedjiesbaai in 2026). The Beach at Caravan Park site is located between Hoedjiesbaai and Bok River

Mouth sites and in 2026 its water quality was also non-compliant, a situation that has persisted over the past 10 years. Previous non-compliance suggests, however, that this site is impacted by poor water quality at neighbouring sites. Although a sustained improvement in water quality has occurred at most Small Bay sites over time, these data indicate that there remains a serious issue of water quality with respect to mariculture operations within areas of Small Bay, which is where the majority of production of the Saldanha ADZ takes place. The prevailing poor water quality in the near-shore waters of Small Bay, particularly in the north/northwest corner, may force sea water abstraction further offshore, or pre-treatment at an increased cost for any land-based fishery and mariculture processing facilities currently operating in the Pepper and Hoedtjies Bay area.

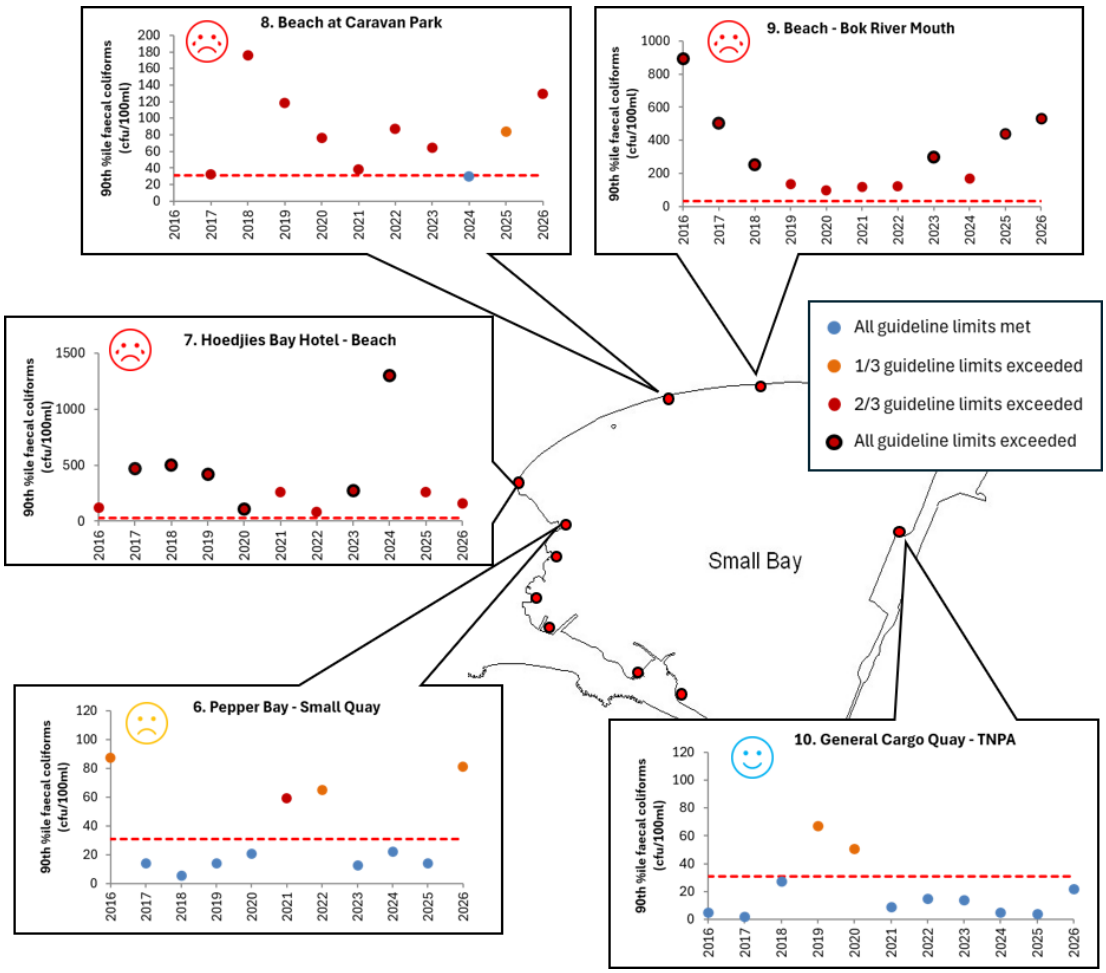


Figure 6.10. Annual 90th percentile faecal coliform counts at five of the 10 sampling stations within Small Bay (2016–2026). Red line = cfu/100 ml guideline limit (DFFE 2022), below which 90% of the count data should fall. Guideline 2: Annual counts median <14 cfu; Guideline 3: not exceed 800 cfu on any one day. All guideline limits met = blue dot, one guideline exceeded = orange dot, two guidelines exceeded = red dot, all limits exceeded = **red** dot black outline.

Faecal coliform counts at three of the four sites sampled within Big Bay in 2026 exceeded the mariculture guidelines, with more than 10% of samples above the acceptable limit (Figure 6.11). This represents a decline in water quality relative to 2025, although Leentjiesklip showed an improvement compared with the previous year.

Historically, water quality in Big Bay has generally complied with the mariculture guidelines since monitoring commenced (Clark et al. 2025), with the exception of Mykonos Harbour, where faecal coliform concentrations have fluctuated and guideline exceedances have occurred intermittently (Figure 6.11). The exposed nature of Big Bay and the resulting high degree of tidal flushing have typically contributed to good microbiological water quality. However, a broader decline in faecal coliform water quality was evident across several Big Bay sites during 2026, including locations that had previously been consistently compliant. Importantly, this non-compliance was not driven by isolated, exceptionally high faecal coliform counts, but rather by persistently elevated concentrations that were only marginally above the guideline threshold. This pattern suggests a sustained increase in background faecal contamination rather than discrete pollution events. The underlying cause is uncertain but may reflect diffuse land-based inputs, increased stormwater runoff, intermittent sewer infrastructure failures, or other catchment-scale influences that temporarily exceeded the assimilative capacity provided by the bay's typically well-flushed conditions.

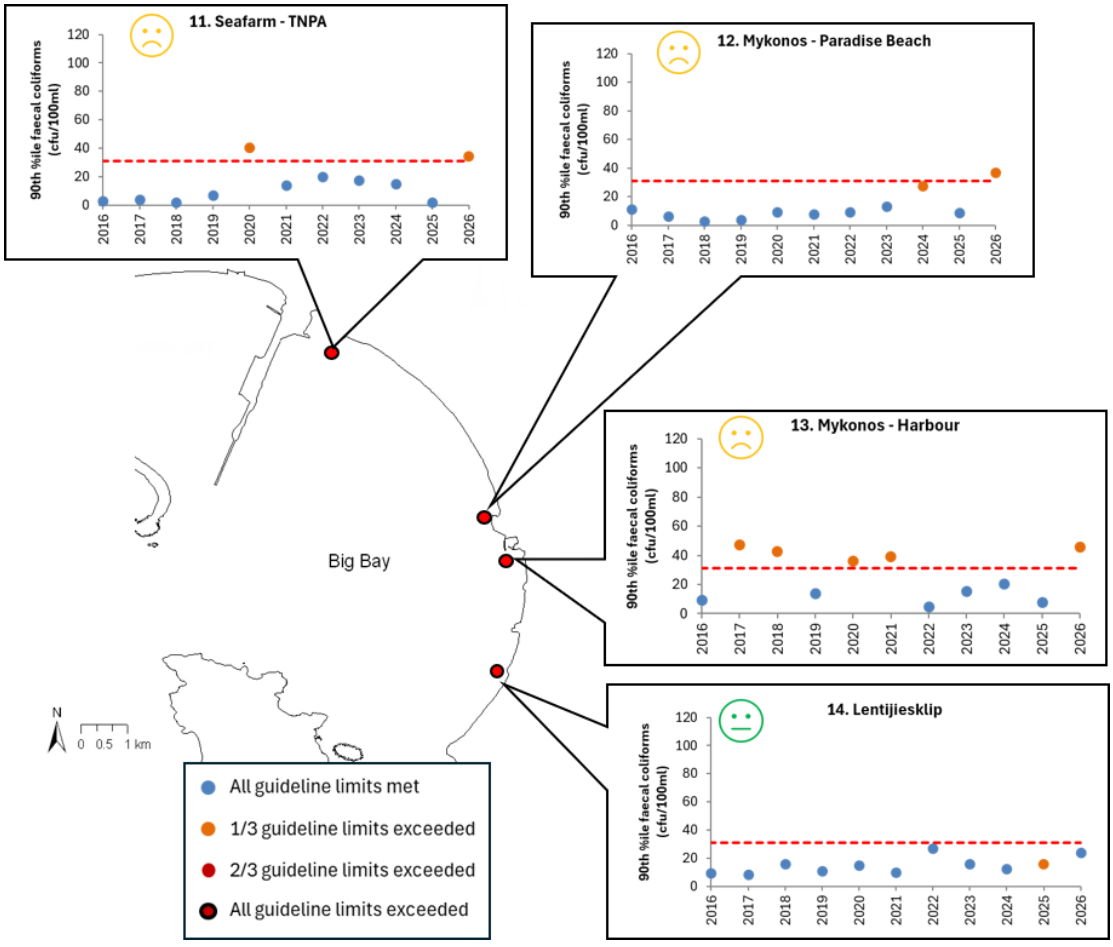


Figure 6.11. Annual 90th percentile faecal coliform counts at four sampling stations within Big Bay (2016–2026). Red line = cfu/100 ml guideline limit (DFFE 2022), below which 90% of the count data should fall. Guideline 2: Annual counts median <14 cfu; Guideline 3: not exceed 800 cfu on any one day. All guideline limits met = blue dot, one guideline exceeded = orange dot, two guidelines exceeded = red dot, all limits exceeded = **red** dot black outline.

6.8 HEAVY METAL CONTAMINANTS IN THE WATER COLUMN

It is common practice globally to monitor the long-term effects of pollution in water bodies by analysing pollutant levels in the tissues of specific marine species or species assemblages (also known as biomonitoring). The extent to which these pollutants are taken up by organisms depends on their bioavailability, that is, the proportion of a contaminant in the environment that is accessible for absorption and accumulation by living organisms. Biomonitoring provides an indication of ecotoxicity and human health risks associated with the consumption of seafood.

Trace metals (or "heavy metals") are often regarded as pollutants of aquatic ecosystems, however, they are also naturally occurring elements, some of which are required by organisms in considerable quantities e.g., copper is often involved in shell formation, nervous system development, and reproduction, particularly in molluscs and echinoderms (Phillips 1980). Aquatic organisms can accumulate these essential trace metals over time. Despite this, all major trace metals do have the potential to be toxic to living organisms above certain 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. 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 1995).

Human activities can greatly increase the rate of mobilization of trace metals from the earth's crust (e.g., by dredging), as well as increase trace metal presence through anthropogenic activity (e.g., industrial outfalls, storm water runoff). Both can lead to increases in the bioavailability of trace metals in coastal waters (Phillips 1980). Analysing dissolved metals in water can be challenging as concentrations are typically low and difficult to detect, 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). For benthic sediments, metals accumulate in sediments over time and typically occur at higher concentrations than dissolved levels, but this still does not reflect their bioavailability (See Chapter 8: Benthic systems). Therefore, analysing metal concentrations in the tissues of aquatic organisms is a 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).

Sessile bivalves (e.g., mussels and oysters) are good indicator species for monitoring water quality as these filter feeding organisms tend to accumulate trace metals, hydrocarbons and pesticides in their flesh. Such sessile molluscs are affected by both short-term and long-term trends in water quality. Monitoring contaminant levels in mussels and/or oysters can provide an early warning of poor water quality and detect changes in contaminant levels in the water column. Therefore, these filter feeding organisms have been successfully used as bioindicator organisms in environmental monitoring programs throughout the world (Kljakovic-Gašpić et al. 2010).

6.8.1 MUSSEL WATCH PROGRAMME

Mussels are abundant in Saldanha Bay, 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 1995, Desideri et al. 2009, Kljakovic-Gašpić et al. 2010).

In 1985 the Marine and Coastal Management branch of the former Department of Environmental Affairs (DEA) initiated the 'Mussel Watch Programme', whereby brown mussels *Perna* sp. or Mediterranean mussels *Mytilus galloprovincialis* were collected periodically from five stations in Saldanha Bay. The programme was discontinued in 2007, and the former SBWQFT re-initiated sampling in 2014 by collecting mussel samples from the same five sites in Small Bay (where trace metal accumulation is of greatest concern) during the annual SOB field surveys. The most recent mussel samples were collected in March 2026 and analysed for the metals lead (Pb), cadmium (Cd), zinc (Zn), copper (Cu), iron (Fe), manganese (Mn) and mercury (Hg) and the metalloid arsenic (As) (Scientific Services (also known as SCISERV) Partial Digestion ICP:OES method). In 2026, duplicate samples were collected from each of the five sites so average values, per site, were used. To facilitate comparison with food quality guidelines, trace metal levels in mussels are presented relative to wet weights of mussel tissue. Data from the Mussel Watch Programme from the past decade are represented in Figure 6.12 to Figure 6.18.

Until 2018, the limits for metals in shellfish as published under regulation R.500 of 2004 were applied to any bivalve samples collected as part of this study. Maximum limits were, however, only specified for cadmium and mercury. In 2018, new South African regulations were published (Regulation R.588 of 2018 in Government Gazette No. 41704). Amendments included a reduction of the acceptable concentration of cadmium in marine mussels from 3 to 2 ppm and the exclusion of mercury from the list, rendering cadmium the only metal for which a legal limit had been stipulated in the South African regulations. For the metals with no specified shellfish limits, national limits as specified for fish and fish products (in the case of arsenic and lead) or those adopted by other countries (in the case of copper and zinc) are applicable to the Mussel Watch Programme. In 2021, international regulations pertaining to metals in foodstuff were again reviewed to set "median guideline limits" for metals in bivalve flesh collected as part of the SOB monitoring programme going forward. In the case of arsenic, where only two values were available, 1 ppm was used (Table 6.5).

Table 6.5. Regulations relating to the maximum levels for metals in bivalves (wet weigh, ppm) in different countries and in South Africa. The revised median guideline limits are indicated in **bold red** text.

Country	Cu	Pb	Zn	As	Cd	Hg
South Africa[1]					2.0	
Canada[2]					2.0	
Australia & NZ[3]		2.0		1	2.0	0.5
European Commission[4]		1.5			1.0	0.5
China[5]		1.5		0.5	2.0	0.5
Revised median guideline limits	**None**	**1.5**	**None**	**1**	**2**	**0.5**

1. Regulation No. 588 of 2018 (Government Gazette No. 41704) published under the Foodstuffs, Cosmetics and Disinfectants Act, 1972 (Act 54 of 1972)
2. Codex. (2019). Codex Standard 193–1995 the general standard for contaminants and toxins in food and feed. Codex Alimentarius Commission. www.codexalimentarius.org which has no limit for zinc.
3. Australia New Zealand Food Standards Code–Schedule 19–Maximum levels of contaminants and natural toxicants. 2021.
4. European Commission 2006. Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. Off J Eur Union, 364(365–324).
5. FAS Beijing Staff. Food and Agricultural Import Regulations and Standards Country Report. 2020. Peoples Republic of China. Network, United States Department of Agriculture Foreign Agriculture Service and Global Agriculture Information Network. FAIRS Annual Country Report. Report Number: CH2019-0198.

With the recent revision of the maximum concentrations of lead permitted in foodstuff by several countries, the median guideline limit has now increased threefold to 1.5 ppm. The European Commission has set a limit (1.5 ppm) and Australia an even higher limit (2 ppm) for bivalves. Mussel samples from Saldanha Bay North (Hoedjiesbaai) have experienced lead concentrations in exceedance of this level between 2016 and 2022 but in the past four years concentrations of lead at this site were lower than guideline limits (Figure 6.12). Lead levels at the Portnet site follow a similar pattern. Mussels collected at the Portnet site have historically had high concentrations of lead in their tissue although values in the last few years remained below the guideline limit (Figure 6.12). The mussel raft and IOT and Fish Factory sites all experienced very low concentrations of lead in 2026, presumably due to better flushing when compared to sites further north in Small Bay. The lead pollution situation in Small Bay is considered to be improving and recorded concentrations in recent years support this, but exceedance at some sites as recently as 2022 suggests that monitoring of lead concentrations at all sites should continue.

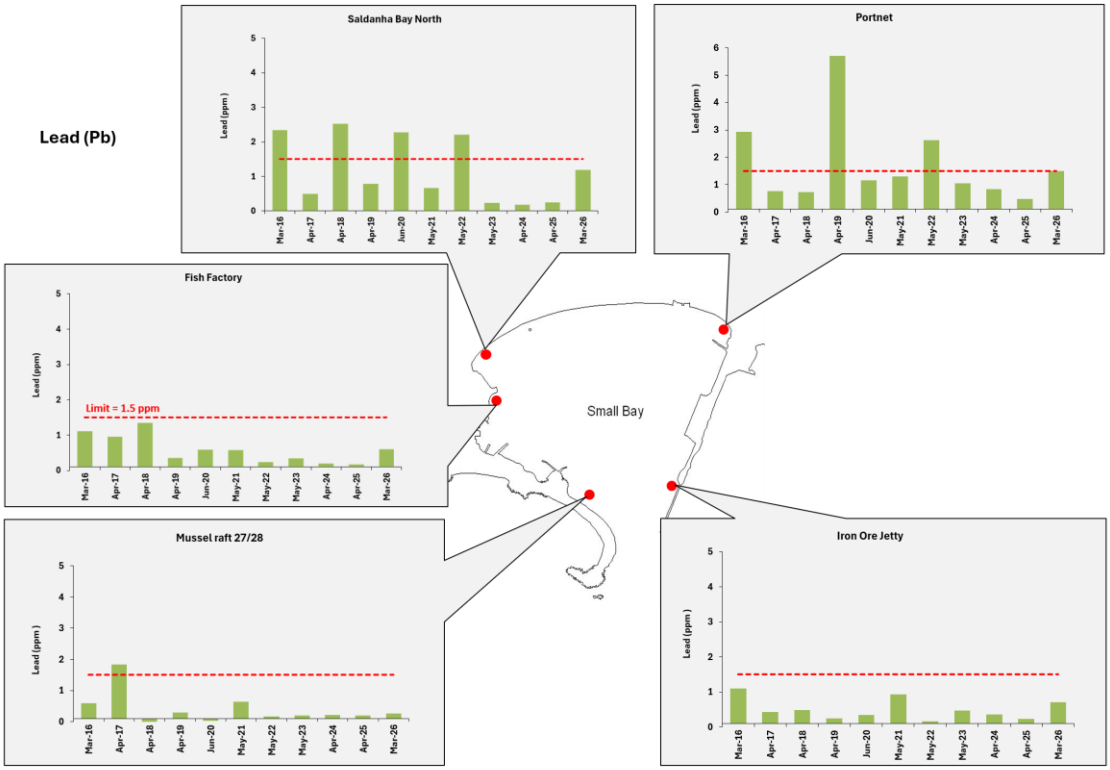


Figure 6.12. Lead concentrations in mussels (wet weight) collected from five sites in Small Bay from 2016–2026 as part of the Mussel Watch Programme. The recommended maximum limit for lead in seafood (1.5 ppm) is shown as a dotted red line.

Cadmium concentrations were very low in all sites, below the 2.0 ppm median guideline limit (Figure 6.13). Cadmium concentrations rarely exceeded the maximum limit of 2.0 ppm, except at the IOT site in 2023, the first exceedance in general since the 2018 sample from the Mussel rafts (Figure 6.13).

Average zinc concentrations at all sites were much lower than any historical regulatory limits e.g., 150 ppm as stated by Canadian Authorities (Figure 6.14). This metal only rose above this limit once at the Saldanha Bay North site (165 ppm in 2016), which was also

elevated in 2019 & 2020 at this site, albeit not above the historical regulatory limit. The recent revision of the maximum concentrations of metals permitted in foodstuff have, however, excluded zinc from this list, meaning there is now no longer a maximum specified limit and the concentrations recorded in Saldanha bay bivalves are unlikely to pose an ecological or human health risk. All recent concentrations of zinc are therefore low and not considered to be of concern.

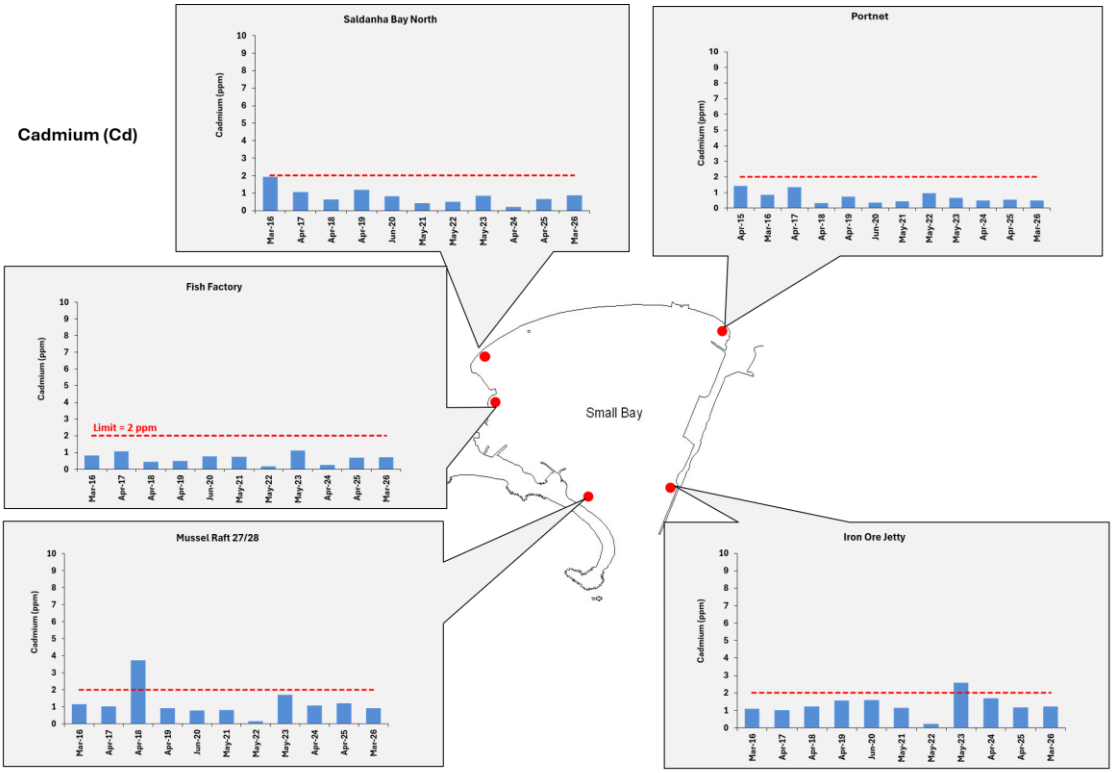


Figure 6.13. Cadmium concentrations in mussels (wet weight) collected from five sites in Bay from 2016–2026 as part of the Mussel Watch Programme. The recommended maximum limits for cadmium are shown by the red dotted line.

Historically, the maximum legal limit of copper in food was listed as 70 ppm in Canada. These limits have since been removed, mainly as copper naturally occurs in high concentrations in the environment and therefore in certain foodstuffs. Copper is widely used in antifouling coatings as it is toxic to marine life at elevated concentrations. Regardless, copper concentrations have never exceeded 5 ppm at any of the sites over the entire sampling period. There appears to be no spatial or temporal trend in level of copper in mussel samples (Figure 6.15). Copper concentrations were not available for 2026.

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 (Brand et al. 1983, Sunda 1989). Historically concentrations were highest at the Portnet site. 2022 saw a peak in concentration at 5.82 ppm (the highest recorded in the Mussel Watch Programme), but the concentrations since then have remained lower. Manganese concentrations

remained relatively high at the IOT in 2025 (Figure 6.16). Manganese concentrations were not available for 2026. Manganese export volume has been steadily increasing to just over 4.5 million tonnes annually 2017–2022. South Africa accounts for approximately 78% of the world's identified manganese resources. 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 South Africa 2017). The MPT is dedicated mostly for export of lead, copper, zinc and manganese concentrates. The manganese loading terminal is midway between the MPT at the base of the IOT, which may influence the relatively high concentrations of manganese seen in mussels from the IOT site.

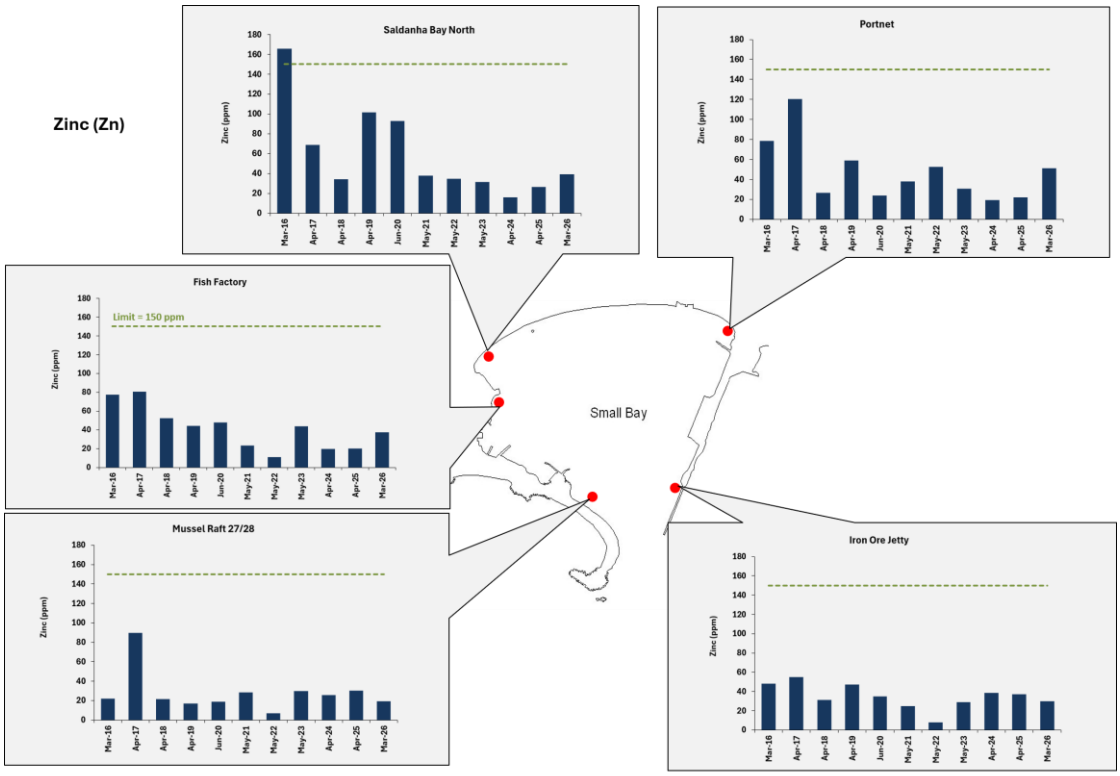


Figure 6.14. Zinc concentrations in mussels (wet weight) collected from five sites in Bay from 2015–2026 as part of the Mussel Watch Programme. No South African limit exists for zinc in bivalves.

Monitoring of concentrations of manganese in mussel tissue should continue as an increase in capacity could lead to an increase in the amount of manganese ore entering the marine environment. In the event that manganese enters the marine environment, there is concern that high levels could be toxic to marine organisms. However, bioaccumulation of trace metals in marine organisms as a direct result of manganese spillage is very unlikely as trace metals usually comprise a very small percentage of manganese ore. With the anticipated increased in manganese storage and handling, and especially considering the high concentrations seen in mussel tissue collected in recent years, measures should be put in place to prevent excessive amounts of manganese dust from entering the Bay.

Arsenic and mercury concentrations within mussel tissues were measured for the first time in 2015 and 2016, respectively. To date, mercury concentrations in mussels have not exceeded 0.2 ppm which is well below the median guideline limit of 0.5 ppm. Mussels collected during the 2026 survey had some of the lowest mercury concentrations on SOB record (Figure 6.17). A comparative study sampled *Mytilus galloprovincialis* mussels from around Saldanha Bay for a year between 2023 and 2024 also reported below guideline concentrations in summer, but in winter Mercury concentrations increased above recommended guidelines (Patel 2025). Mercury (and other heavy metal) concentrations in the tissues of some species have been observed to increase during winter due to factors like dietary shifts, starvation, and reduced growth rates, while warmer temperatures are typically linked to increased growth rates and, therefore, leads to biodilution of trace elements (Jędruch et al. 2019).

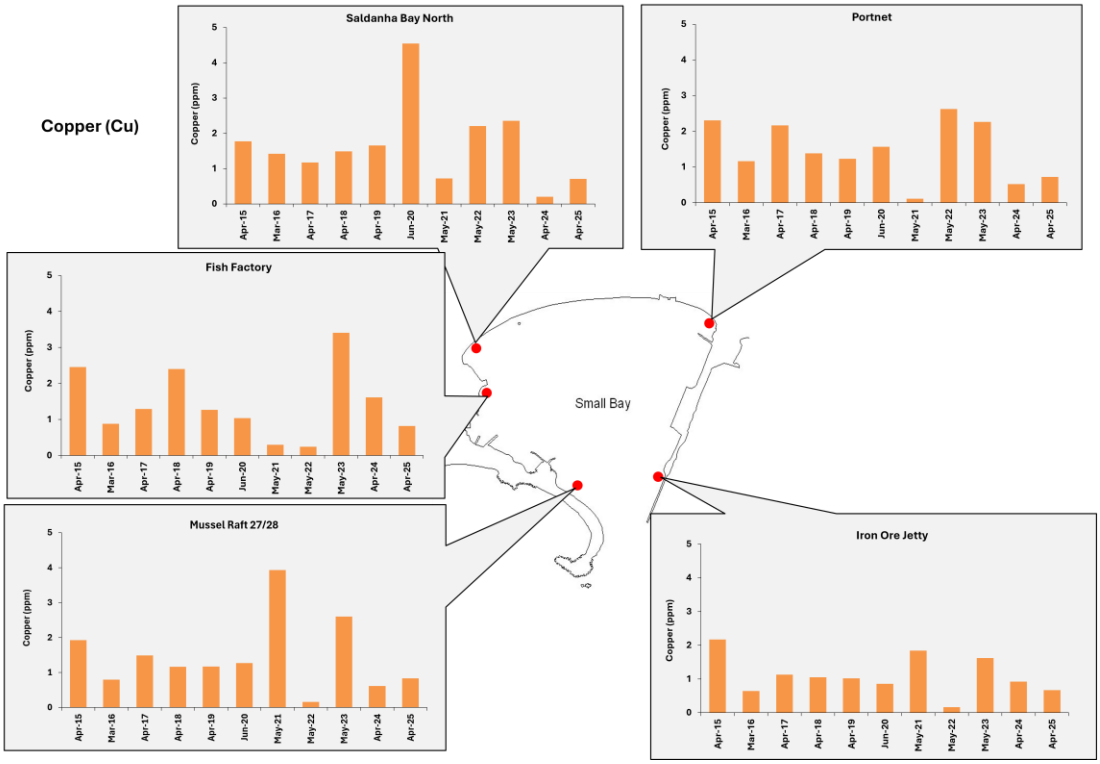


Figure 6.15. Copper concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 2015–2025 as part of the Mussel Watch Programme. The historical maximum limit for copper in seafood is 70 ppm (not indicated on graphs).

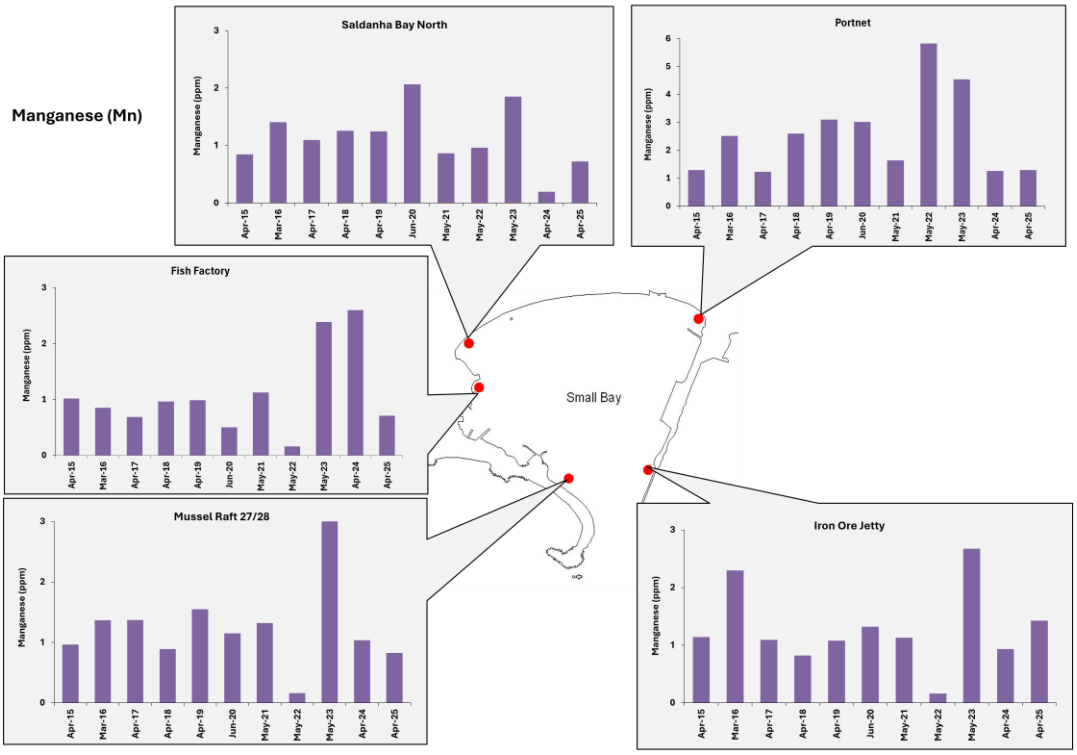


Figure 6.16. Manganese concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 2015–2025 as part of the Mussel Watch Programme. No limits are specified for manganese in seafood.

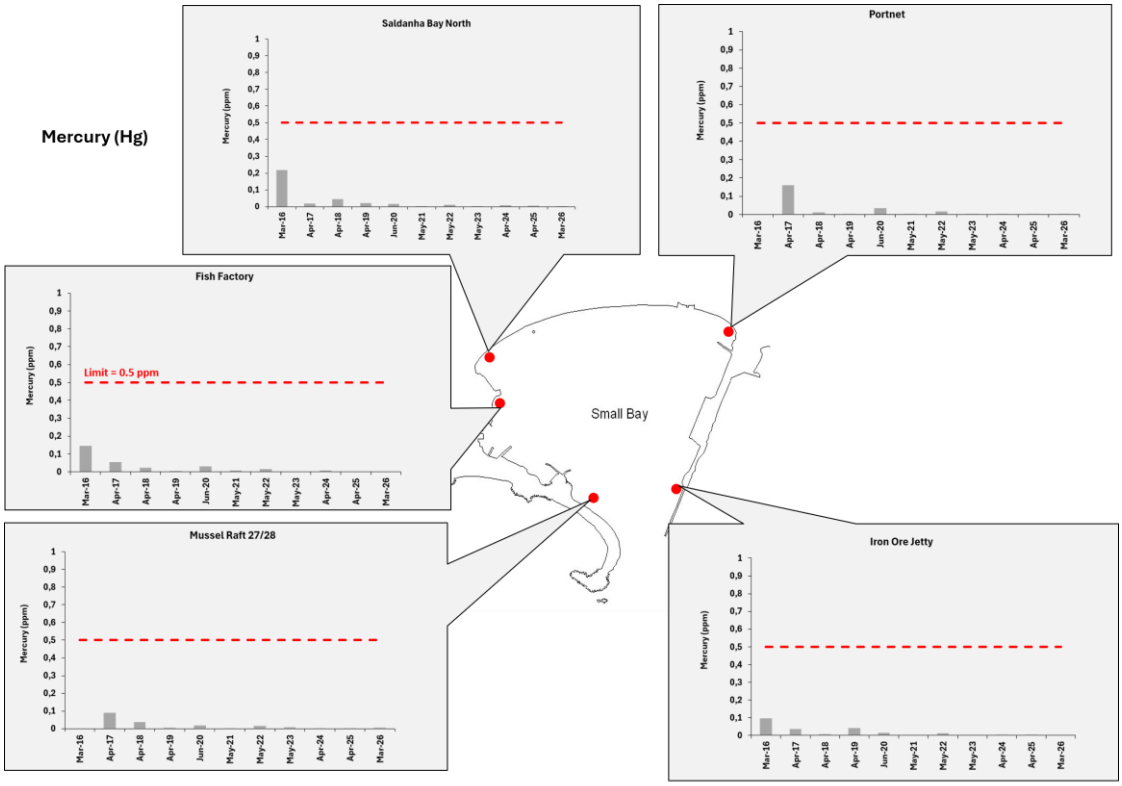


Figure 6.17. Mercury concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 2015–2026 as part of the Mussel Watch Programme. The recommended maximum limits for Mercury are shown by the red dotted line.

Arsenic concentrations in mussel tissue collected from the five sites have fluctuated between 0.5–2.3 ppm. Arsenic levels were always well below the historical threshold of 3.0 ppm. When applying the revised limit, however, levels recorded in 2026 exceeded the acceptable limit of 1.0 ppm at all sites (as they did in 2025), including Saldanha Bay North (Figure 6.18). Arsenic occurs naturally in marine environments, however, anthropogenic sources also contribute to background arsenic levels, include metallurgy (the mining and smelting of metals like lead, copper, and gold); preservation of wood; agriculture (herbicide, fungicide); livestock (sheep dips, feed additives); medicine (human and veterinarian); electronics; cement manufacturing or combustion of fossil fuels and other industry (Fowler 1983, Azcue & Nriagu 1994). Arsenic can bioaccumulate and above certain thresholds is toxic to both marine life and humans.

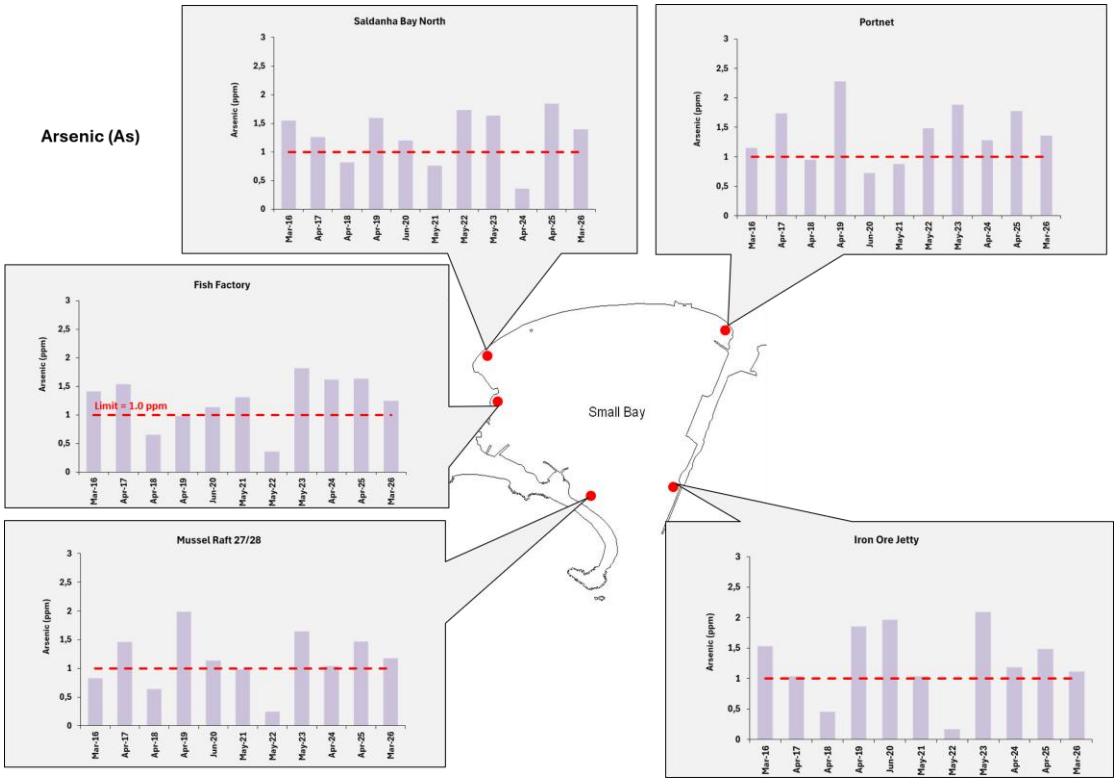


Figure 6.18. Arsenic concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 2016–2026 as part of the Mussel Watch Programme. The recommended maximum limits for Arsenic are shown by the red dotted line.

Background concentrations of arsenic in Saldanha Bay, and in particular within Small Bay, are generally low and fall within ranges considered typical of coastal marine sediments. The SOB surface sediment data, last updated in 2025, showed average arsenic concentrations of approximately 2 ppm in Small Bay, with occasional elevated levels recorded are the SB1 site (see sediment monitoring program) which is near to the mussel watch Fish Farm site. However, concentrations in sediment were consistently below the National Oceanic and Atmospheric Administration (NOAA) Effects Range Low (ERL) guideline of 8.2 ppm for sediments, suggesting a low likelihood of ecological risk. A similar study of average trace metal concentrations in mussel tissue from Saldanha Bay showed that on average arsenic levels were 2.5 ppm which is higher than the temporal

results provided here (Patel 2025). The authors conclude that arsenic levels are influenced heavily by terrigenous inputs (Nekhoroshkov et al. 2024). The Saldanha Bay area is characterized by a lack of the big rivers and developed coastal runoff systems. So it is concluded that the source of arsenic in mussels in the Saldanha Bay area are bottom sediments and groundwater discharges, containing high levels of natural arsenic (as high levels of iron, rubidium, potassium and aluminium further support this) (Nekhoroshkov et al. 2024). Nevertheless, the elevated levels of arsenic in mussel tissue in 2026 are cause for concern and steps should be taken to further identify if this is due to natural background concentrations or of anthropogenic origin (if the latter, management action should be and address potential sources).

6.8.2 MARICULTURE BIVALVE MONITORING

A combined 884 ha of sea space are currently available for aquaculture production in Saldanha Bay. This includes space in Big Bay, Small Bay and Outer Bay North (North Bay). According to the Marine Aquaculture Rights Register, approximately 318.2 ha were leased to 28 individual mussel, oyster, finfish and algae mariculture operators (during 2020, although only 15 of these were actively operational at that time). As of July, 12 farms were considered being active (permits active, not necessarily operational).

South African rights holders engaged in mussel and oyster culture are required to report on trace metal concentrations and bacterial indicators in harvested organisms on an annual basis. The DFFE therefore collects oysters and mussels from the aquaculture farms annually whereafter it is sent to an independent laboratory to test their flesh for four trace metal indicators (lead, cadmium, mercury and arsenic). Data have been obtained for farmed mussels and oysters for the period 2014 to 2026 (Figure 6.19). Gaps in the data exist depending on the frequency of monitoring, the year each company was founded and whether the company was operational in that year or not. For comparative purposes, independent research data from research mussels are also displayed on the graphs. For mussels, this includes data from research surveys by Pavlov et al. (2015) and Bezuidenhout et al. (2015) for the period 2014–2015, and the Mussel Watch Programme by conducted by Anchor (2014–2026). For oysters, data from a monitoring programme initiated by Transnet Port Terminals for the period June 2018 until June 2020 are also included. This programme deployed oyster baskets adjacent to the IOT and MPT. While the SOB samples have only been collected within Small Bay, samples from Pavlov et al. (2015) and Bezuidenhout et al. (2015) were also collected from Langebaan Lagoon and Danger Bay, in addition to Small Bay. Research samples were collected from a variety of locations including the shore, Port (MPT, channel markers), and mariculture infrastructure (mussel rafts, oyster longlines) and were collected close to the surface. Note, for 2024–2026, DFFE provided data from the aquaculture farms, acquired through the PAIA (Promotion of Access to Information Act, 2000) process and therefore individual farm operator information was not supplied (Figure 6.19).

TRACE METALS IN MUSSELS FARMED IN SALDANHA BAY

Previous research has displayed distinct seasonal patterns in trace metal accumulation, with mussels accumulating higher metal concentrations in winter than in summer. This seasonal pattern has been reported in other studies and is thought to be associated with seasonal reproductive patterns, metabolic rate fluctuation and food type and availability associated with upwelling (increased bioavailability of cadmium is correlated with

increased dissolved organic carbon which is elevated when diatom blooms decay) (Sparks et al. 2018). Bezuidenhout et al. (2015) suggested that the observed seasonal dynamics of trace metal concentrations could be a result of the spawning that takes place in summer; with the subsequent large release of gametes (Van Erkom Schurink & Griffiths 1991); effectively eliminating any trace metal accumulation in the gametes from the mussel's bodies.

Lead and arsenic were the most prevalent trace metals in mussel tissue (Figure 6.19). Research mussels typically had higher concentrations of lead and arsenic than those that were farmed. Higher concentrations of cadmium, were, however, periodically reported from farmed mussels in samples collected during 2019–2025 (Figure 6.19).

Since the start of the sampling campaign, concentrations of lead in farmed mussels were generally below the historical guideline limit of 0.5 ppm (Figure 6.19), although mussels from some farms exceeded this limit on occasion. With the recent revision of the maximum concentrations of lead permitted in foodstuff by several countries, a new guideline limit of 1.5 ppm has been adopted. When applying this new limit to both historical and present data, lead concentrations in farmed mussels have rarely exceeded this threshold (only once in 2015, Figure 6.19). The reported concentration of lead in farmed mussels are typically lower than that measured in research samples. Lead concentrations in both research samples and farmed mussels were below the 1.5 ppm limit so far in 2025/2026. Periodic higher concentrations of lead were noted from Small Bay samples compared to elsewhere which further indicates higher lead pollution within Small Bay (Figure 6.19), particularly in nearshore environments that are not well flushed.

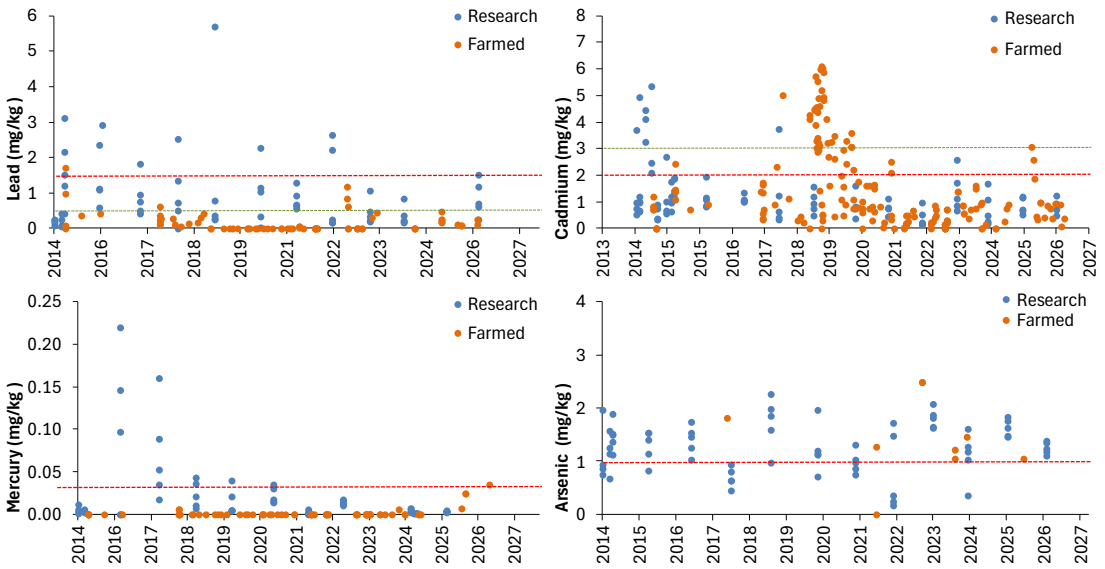


Figure 6.19. Trace metal concentrations (wet weight) in mussel tissue provided by aquaculture facilities ('Farmed') and samples collected by researchers ('Research'), primarily from the shore from 2014–2026, where data were available. The green lines indicate the historical guideline limits applied to samples collected until 2018 for cadmium and 2021 for lead. The red line indicates the revised guideline limits for all metals as applied since 2021.

Data received from mussel farms showed that cadmium concentrations rarely exceeded the prescribed historical limit of 3 ppm (limit applicable until June 2018) (Figure 6.19). In

June 2018, the regulatory limit was decreased to 2 ppm. In samples collected during 2019 and 2020, cadmium concentrations regularly exceeded the limit at aquaculture farms in Outer Bay North, and in two samples from Big Bay and one sample in Small Bay in May 2021 (a notable spike in Cadmium concentration in farmed mussels was observed around 2018/2019). Since 2021 most farmed mussels have reported cadmium concentrations below the prescribed limit, however, in 2025 here were two instances of Cadmium concentrations above the prescribed limit (Figure 6.19). Cadmium naturally occurs in high concentrations within the sediments of near-shore upwelling environments such as the southern Benguela (Griffiths et al. 2005, Summers 2012). High levels of cadmium within the mussels in previous studies have been attributed to disturbances such as dredging, causing trace metals buried in sediment to become re-suspended in the water. The link between cadmium concentration in mussel tissue and dissolved organic carbon associated with decaying diatom blooms described by Sparks et al. (2018), may also play a role in these periodic peaks in cadmium in mussel tissue collected from different localities.

Mercury concentrations submitted to the DFFE have always been within the regulatory limit of less than 0.5 ppm. Since 2014, no exceedance has been recorded and 99% of samples collected contained less than 0.02 ppm of mercury (Figure 6.19). Research samples have also all been below the prescribed limit.

Mussel samples from aquaculture farms were analysed for arsenic for the first time in 2011/2012. For the purpose of this study, a median guideline limit of 1.5 ppm for arsenic (based on that adopted for other countries) in shellfish has been applied to samples collected since 2021. This value was based on the limit for arsenic in fish as published in Regulation R.500 of 2004. Of the aquaculture farms assessed over this period exceedance of the concentration limit occurred several times, most recently in 2025 (Figure 6.19). Mussel tissue has been consistently tested for arsenic at all sites sampled for research since 2013 (Figure 6.19). In 2024, four research samples had arsenic levels above 1 ppm (Figure 6.19).

Mussels are filter feeders which extract particulate matter out of the water column for food; therefore, it is expected that organisms filtering clean water advected into the Bay from offshore will accumulate fewer toxins than mussels filtering potentially contaminated water close to the shore. The reasons for the lower concentrations of trace metals in farmed mussels compared with those on the shore may also be linked to the different depths at which these mussels occur and are collected. Mussels collected during research surveys are collected from intertidal zones close to the surface and are therefore exposed to air and other stressors for part of the tidal cycle. Farmed mussels, on the other hand, are permanently submerged several meters below the surface. This allows them to feed for extended periods of time per day, resulting in higher growth rates. The availability of phytoplankton in deeper areas of the Bay may also facilitate faster growth rates. Faster growth results in less time for the accumulation of toxins within the mussel tissue over the lifetime of the animal and this is the most likely explanation for the observed differences between research and farmed mussel trace metal concentrations.

TRACE METALS IN OYSTERS FARMED IN SALDANHA BAY

As no regulatory limits for metals in oysters have been implemented in South Africa, the median guideline limits for mussels were applied. Lead concentrations in farmed oyster tissue from both Small Bay and Big Bay occasionally exceeded the historical guideline

value of 0.5 ppm, most recently in 2023 (Figure 6.20). Research samples collected as part of the Transnet Port Terminals Monitoring Programme over the period 2018 to 2020, from locations much closer to ore loading facilities than the mariculture farms, also show compliance with guideline levels (Figure 6.20). When applying the revised median guideline limit of 1.5 ppm, all samples, both research and farm samples, fall within the limits, except for the one farmed sample in 2014. As with mussels, oysters collected from farms typically had lower concentrations of lead than oysters sampled for research.

A large number of the samples collected from the farms prior to 2017 would not have met the revised 2018 guideline for cadmium (2 ppm). Farmed and research oyster samples collected from 2017 onwards have nearly all met the revised cadmium guideline, with just two samples exceeding the limit (Figure 6.20). Cadmium concentrations in all research samples collected from Small Bay over the period 2018 to 2020 fell below the 2 ppm guideline, while one of the research samples from Big Bay exceeded the guideline. In 2025/2026, all farmed oysters sampled and tested had cadmium concentrations below 2 ppm.

Mercury concentrations in farm and research samples have all been within the regulatory limit of less than 0.5 ppm since 2014 (Figure 6.20).

Historically, many of the research samples exceeded the regulatory limit for arsenic (1 ppm) (Figure 6.20). As the Transnet Port Terminals oyster monitoring programme was discontinued in June 2020, no oyster samples were analysed for arsenic until 2021 when farmed samples were analysed. Four of the five samples collected in these years exceeded arsenic concentration limits including the solitary sample tested in 2026.

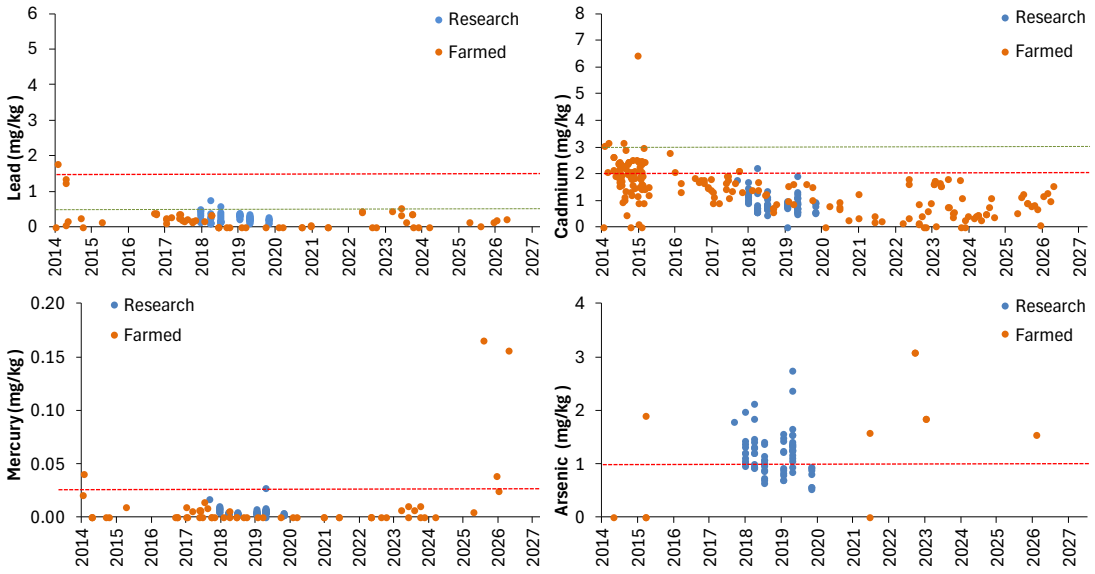


Figure 6.20. Trace metal concentrations (wet weight) in oyster tissue provided by aquaculture facilities and the Anchor Oyster Monitoring Programme (indicated by triangles and circles, respectively) from 2009–2026 where data were available. The green lines indicate the historical guideline limits applied to samples collected until 2021 for lead. The red line indicates the revised guideline limits for all metals as applied since 2021.

In general, trace metal concentrations in farmed oyster samples have largely met the historical regulatory limits for the four trace metals tested, with high levels of compliance in samples collected since 2016. Oysters farmed in Saldanha Bay accumulate trace metals in their tissues at lower levels than mussels, but where occasional exceedance is observed, it is for the same three trace metals, namely arsenic, lead and cadmium, that were also most problematic in mussels. Some concern should be noted here that sampling frequency of farmed oysters recently may be an issue, with, albeit just one, the most recent sample exceeded the limits for arsenic concentration.

Small Bay oyster cages

7 COASTAL EROSION

7.1 INTRODUCTION

Coastal systems worldwide are highly dynamic environments that continually respond to hydrodynamic and anthropogenic variables such as changes in waves, tides, currents, sediment supply, sea level and human activities (López et al. 2017, Theron 2024). These changes can cause coastal erosion, where sand is lost and the shoreline moves further inland, or accretion, where sand builds up and the shoreline moves further towards the sea (Marfai et al. 2008). These changes can happen quickly during storms or develop gradually over many years or decades (Luijendijk et al. 2018). As climate change accelerates and sea levels continue to rise, the risks to coastal stability are becoming more pronounced causing the need for constant monitoring and focused intervention in high-risk areas (Prasad & Kumar 2014, Alves et al. 2020).

In South Africa, coastal environments make substantial contributions to recreation, biodiversity conservation and economic activities such as tourism, fisheries and maritime transport. Tourism alone accounted for approximately 8.9% of the country's Gross Domestic Product (GDP) in 2025, with a considerable proportion of this activity associated with coastal attractions, beaches and related service(Stats SA 2025). However, many beaches along the Western Cape have experienced long-term erosion. This is particularly noticeable along the eastern side of Saldanha Bay, between Spreeuwalle and the entrance to Langebaan Lagoon. This erosion has persisted for at least three decades, with more than 100 m of beach lost at some locations since 1960 and shoreline retreat of up to 40 m recorded in certain areas over a single decade (Flemming 1977).

7.1.1 HISTORY OF COASTAL EROSION IN SALDANHA BAY

Over the past few decades, growing concern has emerged regarding coastline instability in Saldanha Bay, primarily linked to long-term hydrodynamic alterations resulting from the construction of the causeway between Marcus Island and the mainland, along with the Iron Ore Terminal (IOT) in the early 1970s (Clark et al. 2024). These structures have disrupted natural circulation patterns and sediment transport processes, contributing to visible erosion along beaches such as Paradise, Calypso, Leentjiesklip, and Langebaan beaches (Wiese 2013).

It has been proposed that the construction of the Marcus Island causeway and IOT have also modified the bathymetry of Big Bay enough to have an impact on the depositional processes of the beaches. Prior to this construction in 1973, the hydrodynamics of Saldanha Bay was primarily influenced by cyclic natural tidal processes and wave action, thereby sustaining a relatively "stable" bathymetric profile (Flemming 1977, Wiese 2013). The predominant flow direction of the Bay was clockwise, which transported sediment from the north of the Bay to the beaches at the mouth of the Lagoon.

The construction of the causeway and dredging for the IOT drastically modified bathymetry and split Saldanha Bay into two areas (Small Bay and Big Bay), each with their own unique hydrodynamic characteristics (Henrico & Bezuidenhout 2020, Clark et al. 2024). During dredging operations to build the causeway, a large part of the ebb-delta in

front of the mouth of Langebaan Lagoon was removed (Figure 7.1). The loss of the sediment from this location changed the wave refraction pattern causing more wave energy to be directed towards the beach at Langebaan village (Wiese 2013, Flemming 2016) (Figure 7.1).

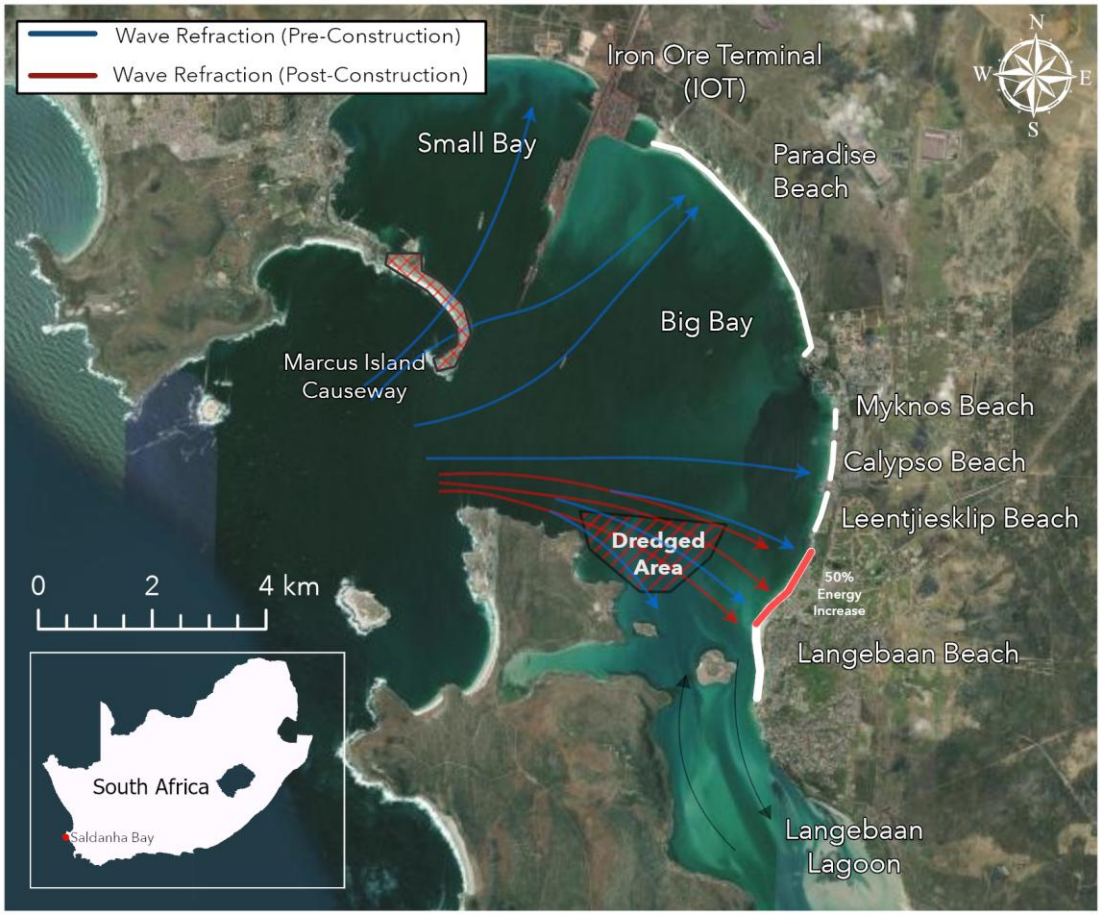


Figure 7.1. Wave refraction patterns of the bay pre-construction of the IOT and Marcus Island Causeway adapted from Flemming (2016). The sediment removed by dredging to create the causeway between Hoedjiespoint and Marcus Island is shown in red hash.

The alteration of Saldanha Bay's hydrodynamics due to causeway construction and dredging also induced significant erosion as a result of increased energy in Big Bay along the beaches near Langebaan town centre, resulting in the loss of two beachfront homes following an extreme weather event in 1997. This prompted emergency interventions, including the installation of ~1.5 km of rock revetment along Langebaan Beach by 2003, which required ongoing reinforcement due to persistent wave action. Between 2003 and 2007, two groynes composed of thousands of geotextile sand containers were constructed to protect the coastline; however, storm damage and public access have caused deterioration, especially above the low-water mark. Similar erosion control efforts included rock revetments near the Alabama Street slipway to protect municipal infrastructure and gabion walls at Paradise Beach to stabilise the coastline, maintained by the local community. Continued erosion led to the drafting of two Environmental Maintenance and Management Plans in 2013: one focused on the eroding beaches at

Langebaan, and the other on coastline areas near Leentjiesklip, the Alabama Street slipway, and Melck Street steps (Common Ground Consulting 2013a b).

Long-term monitoring efforts from 1994 to 2019 by the former SBWQFT (now WQT) have provided valuable insight into the complex seasonal patterns of erosion and accretion along Langebaan's beaches. These studies reveal that erosion typically occurs in winter on the northern section of Langebaan Beach (Figure 7.2,10A1–2A) while the southern section generally erodes in summer (Figure 7.2, 10A2–14). The installation of revetments and groynes significantly influenced coastline stability, with groyne construction (2003–2007) promoting temporary accretion and reducing the magnitude of overall coastline change. However, persistent variability and recent erosive trends highlight the necessity of continued and enhanced monitoring. Given that in situ monitoring comes at a high-cost due to the time and resources required, the introduction of remote-sensing driven coastline monitoring in the State of the Bay (SOB) 2024 report marked a significant advancement for long-term monitoring efforts (Clark et al. 2024). Now and in future, the ability to identify key areas of concern as well as investigate sub-annual coastline variability is critical for remote sensing efforts aiming to achieve the same level of detail as in situ monitoring.

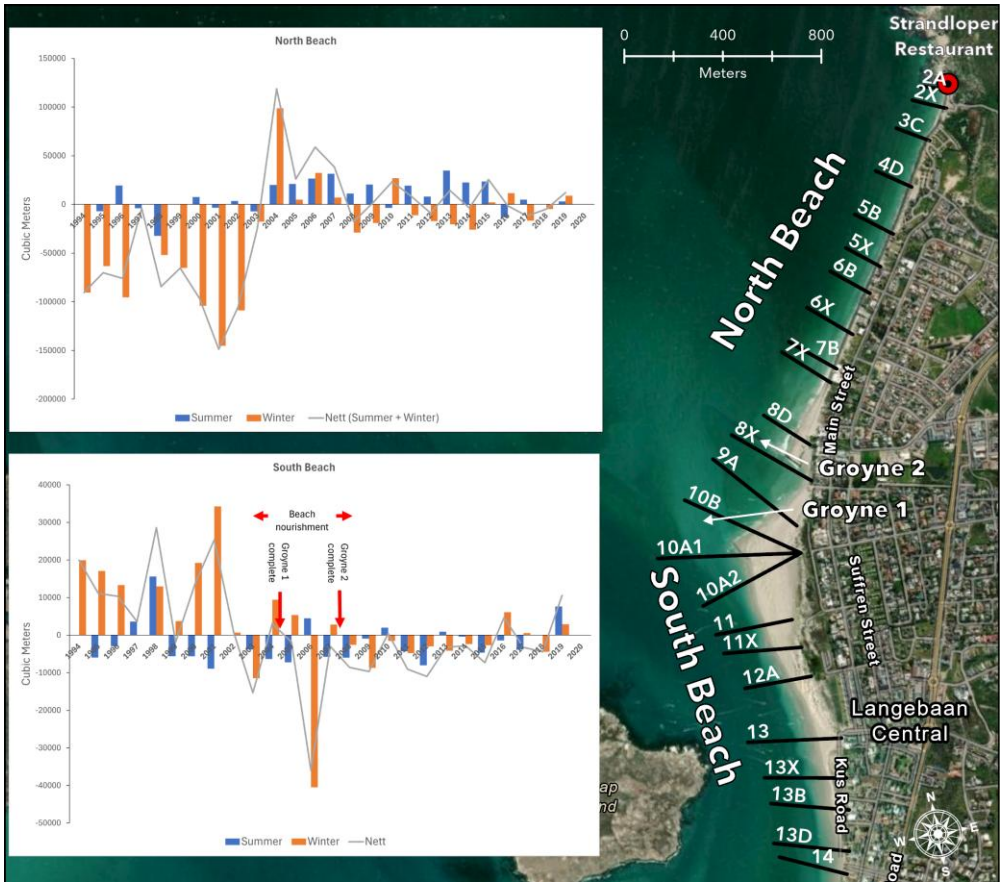


Figure 7.2. Erosion and accretion monitoring of Langebaan Beach (between Leentjiesklip 1 (Strandloper restaurant) and end of Kus Road). Net sand accretion and erosion on Langebaan North Beach (top, the section between the Strandloper restaurant and groyne 1) and South Beach (bottom, south of groyne 1) are shown for summer (blue bars), winter (orange bars) and nett change for the year (grey line), between October 1994–2019. Note that no data were collected in summer 2018. The black lines represent transect measurements done by the monitoring effort (Data Source: SBWQFT (2019)).

7.1.2 Evolution of the Coastal Monitoring Program (2024–2025)

Within the previous two years of coastal monitoring undertaken as part of the SOB reports, historical shoreline dynamics in Saldanha Bay were investigated through the application of progressively more advanced remote sensing and geospatial analysis techniques.

2024 MONITORING PROGRAM

The 2024 monitoring programme marked the transition from traditional field-based shoreline surveys to a more cost-effective, remote sensing approach. The primary objective was to determine whether satellite-derived shorelines could provide reliable long-term monitoring of coastal change within Saldanha Bay.

The Shoreline Analysis and Extraction Tool (SAET) were selected as the initial assessment tool due to its ability to automatically extract shoreline positions from satellite imagery and evaluate changes over extended periods. The satellite-derived shoreline dataset was first used to calculate net shoreline change across all monitored beaches, providing an initial spatial assessment of where the coastline had experienced overall erosion, stability or accretion from 1985–2024. These results were then supported by Linear Regression Rate (LRR) analysis on the northern and southern sections of Langebaan main beach, which quantified the average rate and direction of shoreline change over time. The calculated shoreline change rates were subsequently used to develop preliminary projections of future shoreline positions, providing an indication of how existing trends may continue if similar coastal conditions persist.

The results supported previous in situ monitoring undertaken by the SBM and WQT, confirming that shoreline erosion decreased considerably following completion of the groyne system in 2007. Following construction, sediment began to accumulate around the groyne structures, contributing to the stabilisation and widening of adjacent sections of the beach. The analysis further demonstrated that these longer-term shoreline trends could be monitored remotely at a lower cost and with greater temporal coverage than conventional in situ monitoring alone.

However, limitations were identified in that SAET provided useful regional-scale shoreline mapping but required considerable manual quality control, struggled under variable environmental conditions, and lacked the flexibility required for detailed seasonal investigations. These limitations motivated the evaluation of alternative shoreline extraction methods for future monitoring.

2025 MONITORING PROGRAM

Building on the proof-of-concept established in 2024, the 2025 monitoring programme focused on improving shoreline detection accuracy and expanding the analytical capability of the monitoring framework.

Following the evaluation of multiple shoreline extraction tools, CoastSat was selected as the preferred monitoring tool because of its improved shoreline detection algorithms, automated processing workflow, and ability to analyse extensive satellite archives (Vos et al. 2019b). More than 600 cloud-free Sentinel-2 images (2015–2025) were processed to generate a consistent shoreline dataset for Saldanha Bay. LRR analysis was then

applied across all monitored beaches to quantify decadal shoreline change rates and establish the spatial distribution of erosion and accretion throughout the study area.

These shorelines change rates were subsequently incorporated into a modified Coastal Vulnerability Index (CVI), where the LRR analysis was used to classify different sections of the coastline according to their relative vulnerability. This provided an objective method for distinguishing areas undergoing erosion from those experiencing accretion, and enabled priority locations to be selected for more detailed temporal investigation. Two primary areas were identified around the Langebaan groyne system: an accretion hotspot adjacent to the northern groyne and an erosion hotspot between the two groynes. Seasonal analyses revealed that these hotspots followed distinct annual sediment cycles. The accretion hotspot experienced sediment accumulation during summer (approximately 16 m/year) followed by a period of moderate erosion in winter. Conversely, the erosion hotspot experienced average shoreline retreat of approximately 3 m/year, with erosion greatest over winter months and partial recovery by form of deposition during summer.

These findings demonstrated that the groynes were found to slow the rate of erosion surrounding the key areas of Langebaan beach but were to be a temporary solution and had not been found to stabilise the entire coastline with continues erosion between the structures. Continued monitoring was therefore required to determine whether the structures remained effective over time, while also tracking the progression of erosion in adjacent and intervening areas that received limited protection, such as between the structures and along Paradise Beach. This was necessary to establish whether shoreline conditions were stabilising or deteriorating and to identify areas where additional management or intervention might be required.

7.1.3 Hydrological Factors

Although the modified CVI was effective in 2025 in identifying areas of potential concern, an assessment based solely on shoreline change could not fully explain the processes driving coastal variability. The 2025 report therefore recommended expanding the CVI to incorporate additional physical parameters, including the rate of sea-level change (mm/year), coastal elevation (m), coastal slope (%), nearshore bathymetry (m), mean tidal range (m), significant wave height (m), beach width (m), and geomorphological characteristics (Gornitz 1991). Of these variables, significant wave height was selected as the first to be incorporated into the modified CVI due to the abundance of wave data generated by studies examining changes in the hydrodynamics of Saldanha Bay since the construction of the Causeway and IOT (Flemming 1977, 2016, Luger et al. 1999, 2006, Gericke 2008, Wiese 2013, Henrico & Bezuidenhout 2020, Du Toit et al. 2022).

The 2024 monitoring programme introduced the potential influence of these hydrodynamic changes by examining long-term variability in the regional wave climate. Analysis of computer-modelled wave data, together with measurements collected by a wave-monitoring buoy, revealed that the frequency and magnitude of high significant wave heights (SWH) has increased every decade since the 1960s. These findings suggested that the wave-energy regime influencing the Saldanha Bay coastline may have intensified over recent decades and highlighted the need to investigate whether changes in wave conditions were directly associated with the observed patterns of shoreline variability and what areas were of most concern.

Building on these findings, the 2025 monitoring programme further investigated the relationship between wave climate and shoreline variability, with particular emphasis on seasonal behaviour at identified erosion and accretion hotspots. Both hotspot areas along Langebaan Beach displayed a distinct seasonal shoreline cycle, characterised predominantly by erosion during winter and accretion during summer. This pattern suggested that elevated winter wave energy and storm activity may contribute to seasonal sediment removal and eroding shoreline movement, while lower-energy summer conditions allowed sediment to return to the beach and promoted shoreline recovery. However, the extent to which seasonal changes in wave conditions directly control shoreline behaviour has not yet been statistically quantified.

7.2 METHODOLOGY

7.2.1 DATA SOURCES

This report utilised freely available satellite imagery from the Sentinel-2 constellation to support the coastal monitoring campaign. Sentinel-2 provides multispectral imagery across 13 spectral bands, with spatial resolutions of 10 m, 20 m, and 60 m depending on the band (ESA 2015). The visible and near-infrared bands at 10 m resolution enable the accurate identification of shoreline positions and the detection of changes in coastal morphology over time.

Tidal observations used to correct shoreline positions were obtained from the South African Navy Hydrographic Office (SANHO) (SANHO 2026). These data were used to adjust the shoreline to account for the differences in sea level caused by the tides at the time of satellite image acquisition.

Significant wave height data were obtained from the European Centre for Medium-Range Weather Forecasts Reanalysis v5 (ERA5) dataset through the Copernicus Climate Data Store (Hersbach et al. 2023). These data were used to characterise wave conditions within the study area and support the interpretation of observed shoreline changes.

7.2.2 COASTSAT

Coastal erosion modelling for this report was undertaken in the CoastSat workflow, which is designed to characterise shoreline variability, enabling standardised investigations into shoreline responses to natural erosion, and other coastal phenomena (Vos et al. 2019a). CoastSat is an open-source Python toolkit that enables automated extraction of shoreline positions from more than three decades of publicly available satellite imagery. A defined region of interest is first created around the study area and used to query satellite image collections through the Google Earth Engine, accessing satellite imagery from Landsat 5, 7, and 8 missions as well as Sentinel-2 (Figure 7.3).

Once retrieved, the imagery is automatically filtered for cloud cover using quality assessment bands and cropped to the selected region of interest while still on the Google Earth Engine server. This step improves processing efficiency and minimizes unnecessary data transfer. Only relevant spectral bands are retained, including the red, green, blue, near-infrared, and shortwave infrared (SWIR) bands. Pre-processing continues with pan-sharpening for Landsat 7 and 8 images to enhance spatial resolution from 30 to 15 m using panchromatic fusion. Landsat 5 imagery is down-sampled to 15 m

via bilinear interpolation, while Sentinel-2 images, the 20 m SWIR1 band is down-sampled to 10 m by bilinear interpolation, so that all the bands are at 10 m resolution (Figure 7.3).

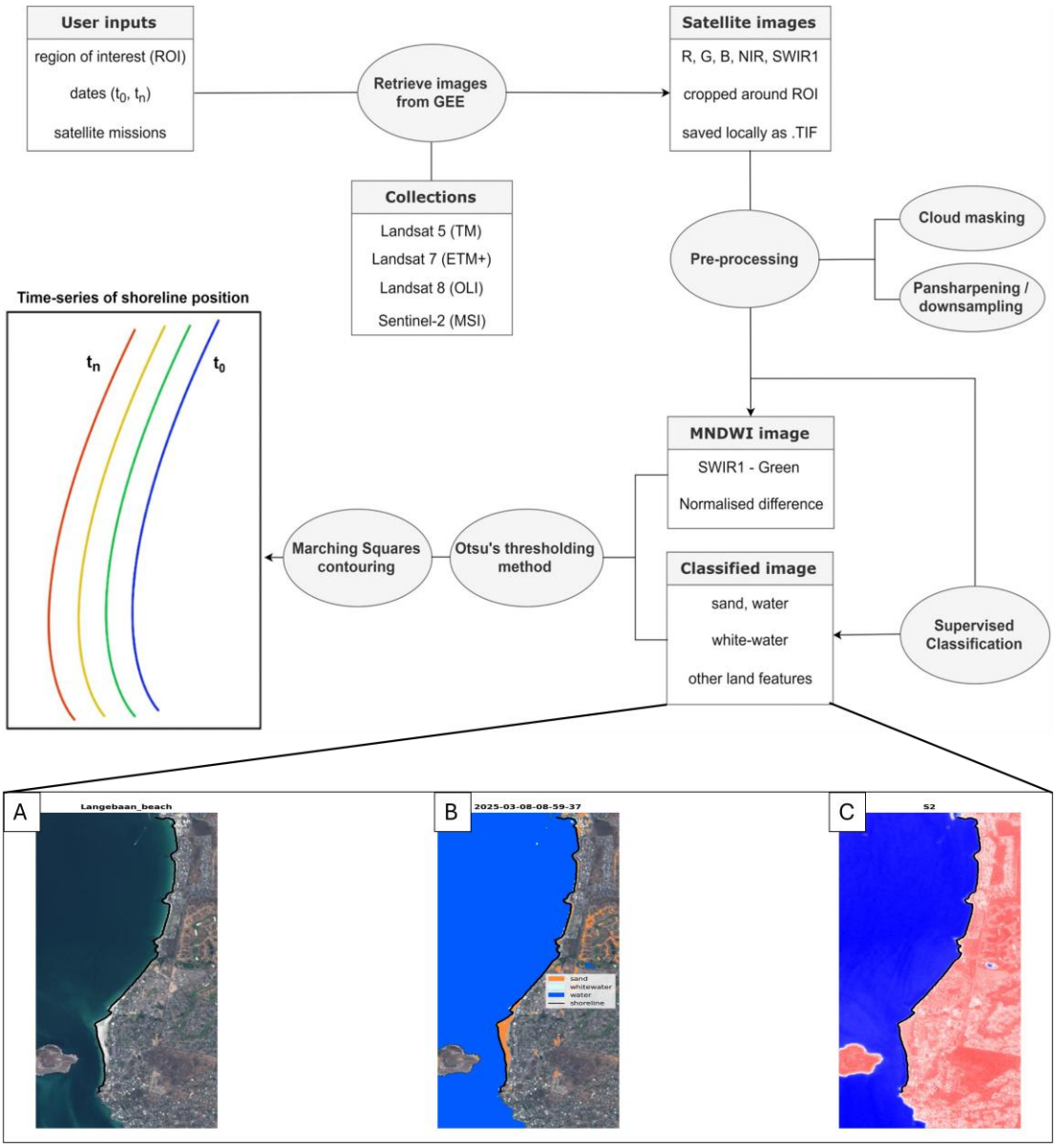


Figure 7.3. CoastSat workflow (Vos et al. 2019a) a) True colour imagery b) Output of image classification where each pixel has been labelled with one of three classes: 'sand', 'white-water' and 'water' (the 4th class 'other land features' corresponds to the unlabelled pixels). c) Example of interest within Red Green Blue (RGB) image located at Schrywershoek in Saldanha Bay, Sentinel-2 (MSI).

Following image pre-processing, shorelines were extracted using the Modified Normalized Difference Water Index (MNDWI):

$$\mathrm{MNDWI} = \frac{\mathrm{Green-SWIR}}{\mathrm{Green+SWIR}}$$

where the green and SWIR bands maximise the spectral contrast between water and land surfaces. Water typically exhibits high reflectance in the green wavelengths and low reflectance in the SWIR wavelengths, whereas dry sand and terrestrial surfaces exhibit relatively higher SWIR reflectance. As a result, the MNDWI enhances the wet-dry shoreline boundary while reducing interference from vegetation, shadows, and wave foam (Vos et al. 2019b).

Otsu's thresholding algorithm (Otsu 1979) was subsequently applied to automatically identify the optimal threshold separating water and non-water pixels from the MNDWI histogram. The resulting binary water masks were then subjected to morphological filtering procedures to remove isolated artefacts and improve shoreline continuity. Finally, shoreline vectors were extracted from the binary masks using a marching square contouring algorithm, allowing sub-pixel interpolation of shoreline positions despite the coarser resolution of the source imagery (Vos et al. 2020).

As the satellite images were acquired at different stages of the tidal cycle, the extracted waterlines did not represent a consistent elevation. Shoreline positions were therefore corrected to a common reference elevation using the water level at the time of image acquisition and the local beach-face slope:

$$\Delta x = \frac{Z_{ref} - Z_{wl}}{m}$$

where Δx is the horizontal shoreline correction, Z_{ref} is the selected reference elevation, Z_{wl} is the water level at the time of image acquisition, and m is the beach-face slope. The calculated correction was applied to each original transect intersection to produce a tidally corrected shoreline position.

The 2025 analysis relied on predictions from the FES2022 global tide model, whereas the present assessment used observations from the SANHO tide gauge in Saldanha Bay (SANHO 2026). These observations were analysed using the T-Tide harmonic analysis package (Pawlowicz et al. 2002), which uses observed tidal patterns to estimate water levels at any point in time. This allowed the local tide level to be calculated for the exact time each satellite image was captured across the full observation period.

Subsequently, spatially varying beach-face slopes were incorporated into the tidal correction. Unlike the 2025 analysis, which applied a single representative beach slope to all transects within each beach, the present assessment estimated a separate slope for each transect using the CoastSat.slope (CoastSlope) extension (Vos et al. 2020). This enabled shoreline corrections to reflect local differences in beach slope rather than assuming a uniform gradient across the entire beach. CoastSlope determines the most likely beach-face slope by analysing the relationship between tidally induced shoreline-position variability and tidal elevation across multiple satellite observations. This approach better represents alongshore differences in beach morphology and has previously been incorporated into the South African shoreline-monitoring framework developed by (Theron 2024).

7.2.3 Correlation Analysis

To assess the influence of the wave regime on surrounding shoreline dynamics, a statistical correlation analysis was undertaken to determine whether seasonal variations in shoreline position were significantly related to changes in wave conditions. The

analysis was conducted separately for each shoreline transect to identify spatial variations in shoreline sensitivity to changing wave conditions.

Due to the inability to obtain the infield WaveRider buoy data for the study period, ERA5 SWH data were used as an alternative source of wave information. Hourly ERA5 SWH data were obtained for the period 2016–2026 (Figure 7.4). The data was extracted from a location approximately 34 km west of Saldanha Bay, as ERA5 data within the immediate coastal zone can be less reliable due to limitations in resolving nearshore processes (Bessonova et al. 2025). Both shoreline position data, available approximately every five days, and hourly SWH data were averaged into seasonal values to examine overall seasonal shifts in shoreline position in relation to changing hydrodynamic conditions.

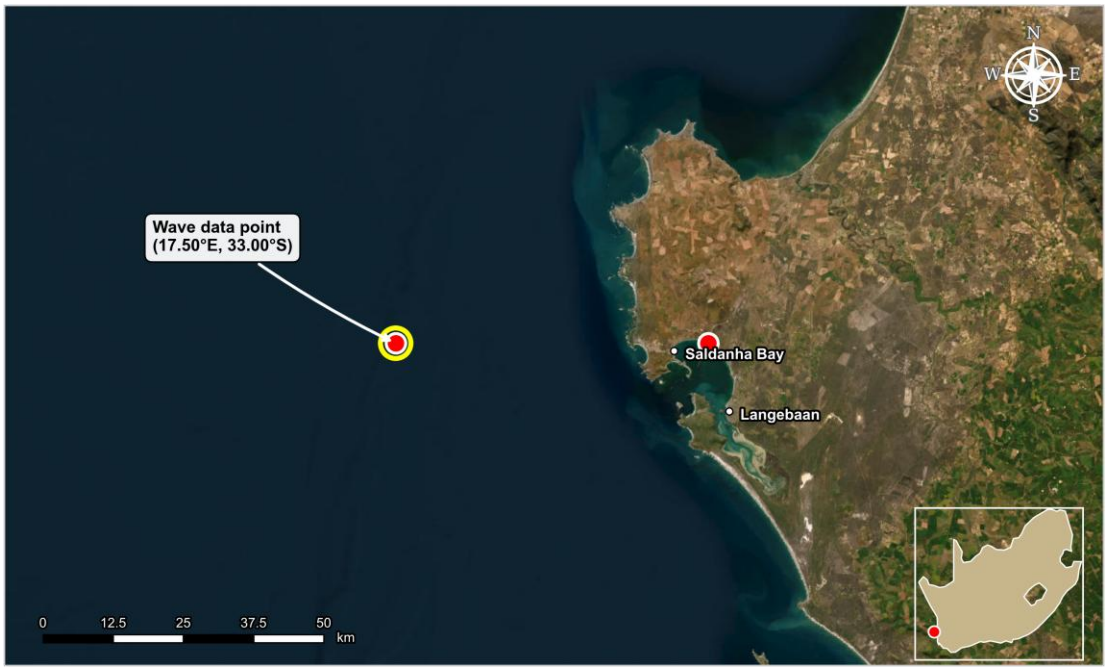


Figure 7.4. Location of the ERA5 grid point selected for the significant wave height analysis.

To determine these overall seasonal shifts, season-to-season changes were then calculated to assess whether the change in seasonal wave conditions was associated with corresponding shoreline movement. Both SWH and shoreline data was calculated as the difference between average values for consecutive seasons, for example the change from December–February (DJF) to March–May (MAM). In this analysis, increasing shoreline change values indicated that the shoreline had moved further away from land, representing accretion of the beach, while decreasing values indicated movement towards the land, representing erosion. As the seasonal-change data showed no substantial departure from a normal distribution, Pearson's correlation test was selected to assess the strength and direction of the linear relationship between changes in SWH and shoreline position.

The Pearson's correlation coefficient was calculated for each transect to measure how strongly seasonal changes in SWH were related to seasonal changes in shoreline position, and whether the relationship was positive or negative on a linear scale. A negative correlation indicated that the variables moved in opposite directions, meaning

that increases in wave height were generally associated with erosion of the shoreline, while a positive correlation indicated the variables moved in the same direction mean that increases in wave height were generally associated with accretion of the shoreline. The coefficient of determination (R^2) was than calculated to determine how much of the variation in shoreline change (positive or negative) could be statistically associated with changes in wave height (Pearson 1909). Finally, a simple linear regression was then fitted to each transect to estimate the amount of shoreline movement associated with a change in SWH, providing an indication of how sensitive different sections of the coastline were to change wave conditions.

The strength of the Pearson correlation was classified using the absolute value of the correlation coefficient, with |r|<0.10 categorized as very weak, 0.10–0.39 as weak, 0.40–0.69 as moderate, 0.70–0.89 as strong, and 0.90–1.00 as very strong (Table 7.1) (Schober et al. 2018).

Table 7.1. Classification of Pearson correlation strength based on the absolute correlation coefficient (|r|) taken from Schober et al. (2018)

Category	Pearson Strength Correlation (r)
Very weak (negligible)	0.00–0.09
Weak	0.10–0.39
Moderate	0.40–0.69
Strong	0.70–0.89
Very Strong	0.90–1.00

Following the overall SWH-shoreline correlation analysis, further assessment was done to determine what the primary wave direction correlated the most to erosion among the identified transects. The directional analysis focused only on sections of shoreline where increases in wave height had already been significantly associated with erosion. Hourly ERA5 mean wave-direction data were then paired with the corresponding SWH data and grouped into eight directional sectors: north, northeast, east, southeast, south, southwest, west, and northwest. The relationship between wave height and shoreline change was then examined separately for each wave direction. For each beach, the dominant erosional wave direction was identified as the direction associated with erosion at the greatest number of transects. Where two or more directions affected the same number of transects, the direction showing the stronger relationship with erosion was selected.

7.3 Results

7.3.1 Annual Coastal Monitoring Update

An annual coastal monitoring update has been undertaken for the key beaches within the study area to continually assess ongoing changes in shoreline position and beach dynamics. A five-year assessment period (2020–2026) was used to assess each beach to determine how shoreline changes observed during the most recent year (i.e., 2025–2026) influence the prevailing running average. This approach provides greater context for interpreting whether recent erosion or accretion represents a short-term fluctuation or a continuation of an established pattern.

LANGEBAAN-MYKONOS BEACHES

Langebaan, Leentjiesklip, Calypso, and Mykonos beaches were moderately changing, with coastline change rates typically ranging between 0 and 1 m/year (Figure 7.5). A notable erosion hotspot (-2.29 m/year) was identified between Langebaan's first and second groynes, while an accretion hotspot (15.8 m/year) was observed just south of the groynes and north of Schaapen Island (Figure 7.5). The beach opposite Schaapen Island showed repeated reversals in shoreline behaviour, with sections alternating between erosion and accretion over the study period.

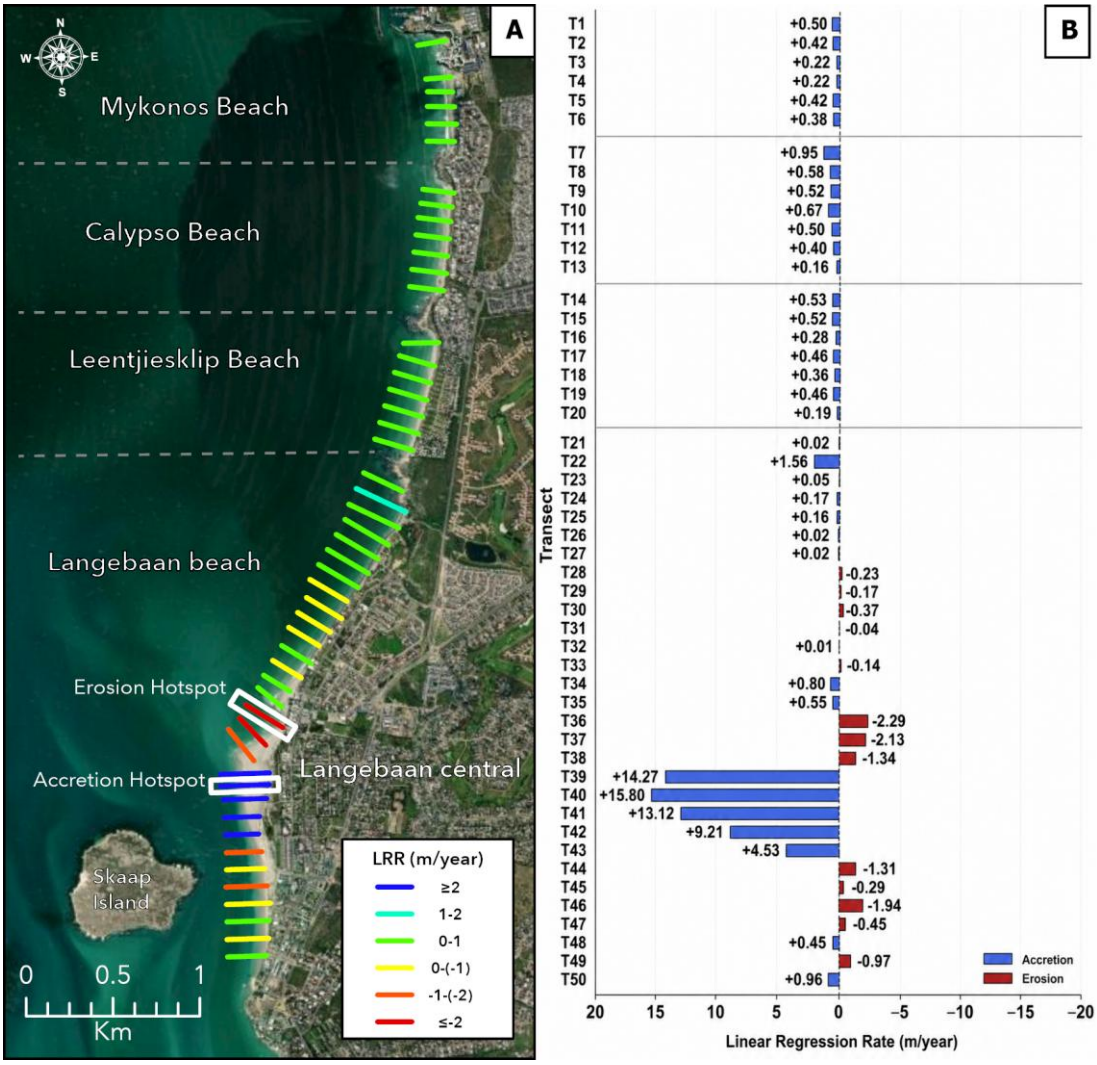


Figure 7.5. Linear Regression rates over 5 years (A) applied to Langebaan, Leentjiesklip, Calypso and Mykonos Beach, and (B) from baseline (0 m/year) at every 100 m transect along the coast.

The primary accretion and erosion hotspots were further investigated using seasonal shoreline position analysis to determine how shoreline movement varied throughout the year and determine if any consistent seasonal patterns of erosion and accretion were present. For each transect, the shoreline position was calculated based on its distance from a fixed reference baseline (Figure 7.6). These distances from baseline were then averaged according to seasonal periods, defined as Summer (December–February),

Autumn (March–May), Winter (June–August), and Spring (September–November) (Figure 7.6).

At the erosion hotspot, the coastline retreated by approximately 2.3 m/year over the past five years (Figure 7.6). Seasonal trends indicate sediment accumulation at the start of spring and summer, followed by erosion during autumn and winter. This results in an average shoreline retreat of approximately 12 m between summer and winter, with a maximum retreat of approximately 16.5 m in 2020 and a minimum retreat of approximately 4.4 m in 2022 (Figure 7.6).

At the accretion hotspot, consistent sediment accumulation averaging around 16 m/year caused seasonal trends to be less pronounced compared to the erosion hotspot (Figure 7.6). However, seasonal patterns were still evident, with most accumulation occurring from summer to autumn and erosion during winter. From the winter of 2023 onward, clearer winter-based erosion has been evident with a more distinguished seasonal cycle of winter erosion and summer accretion over the last year. (Figure 7.6). On average, the shoreline accreted by approximately 6.3 m between summer and winter. The greatest seasonal accretion, approximately 13.8 m, occurred in 2020, while the greatest erosion, approximately 5 m, occurred in 2026.

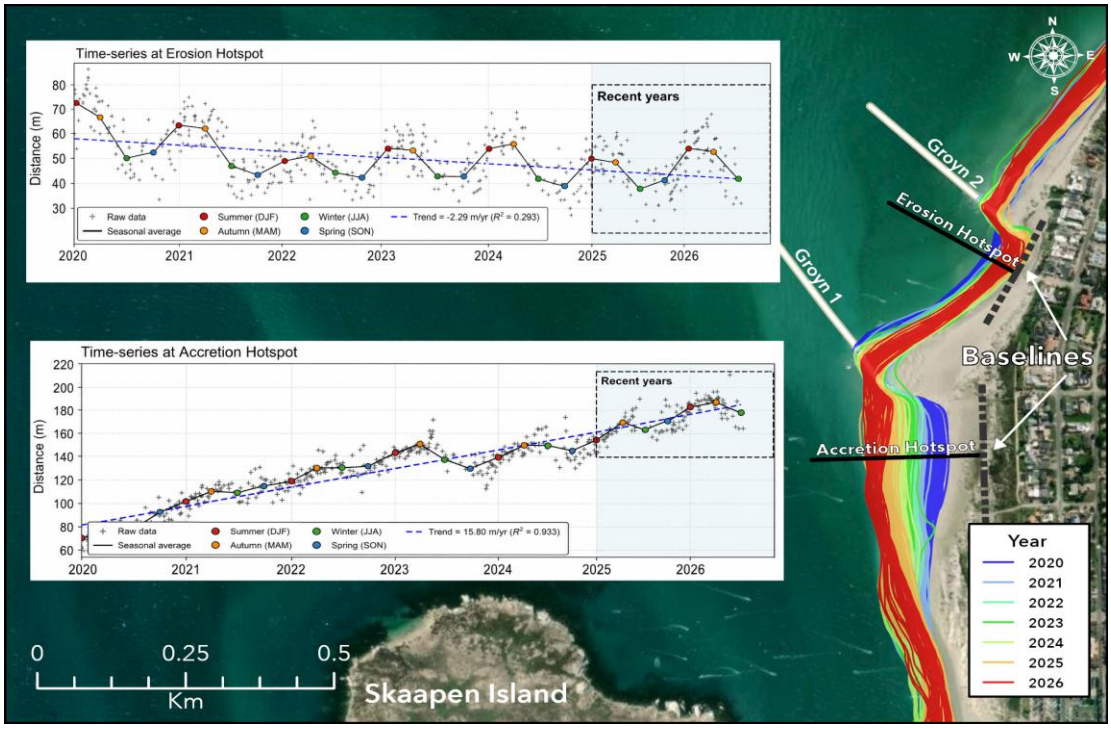


Figure 7.6. Seasonal shoreline changes at Langebaan beach (2020–2026), with transects shown in black, groynes in white, baselines as grey dotted lines and shoreline positions color-coded by year. Time-series plots highlight trends at two key locations, showing raw shoreline positions (+), seasonal averages (Summer, Autumn, Winter, Spring) and linear regression lines (dashed blue).

PARADISE BEACH

The coastline between Paradise Beach and the IOT underwent moderate change during the assessment period, with shoreline-change rates generally ranging between 0–1 m/year (Figure 7.7). Higher rates of accretion were recorded near the IOT, ranging from

approximately 0.4 to 1.6 m/year, while the central sections of the beach (Transects 15 to 27) displayed slower accretion rates of approximately 0.01–0.15 m/year. Localised patches of erosion were observed between Transect 21 and 43, with erosion rates ranging from approximately -0.04 to -0.6 m/year (Figure 7.7). Accretion increased again between Transect 44 and 48 in the vicinity of the gabion walls, where rates reached up to 1.03 m/year.

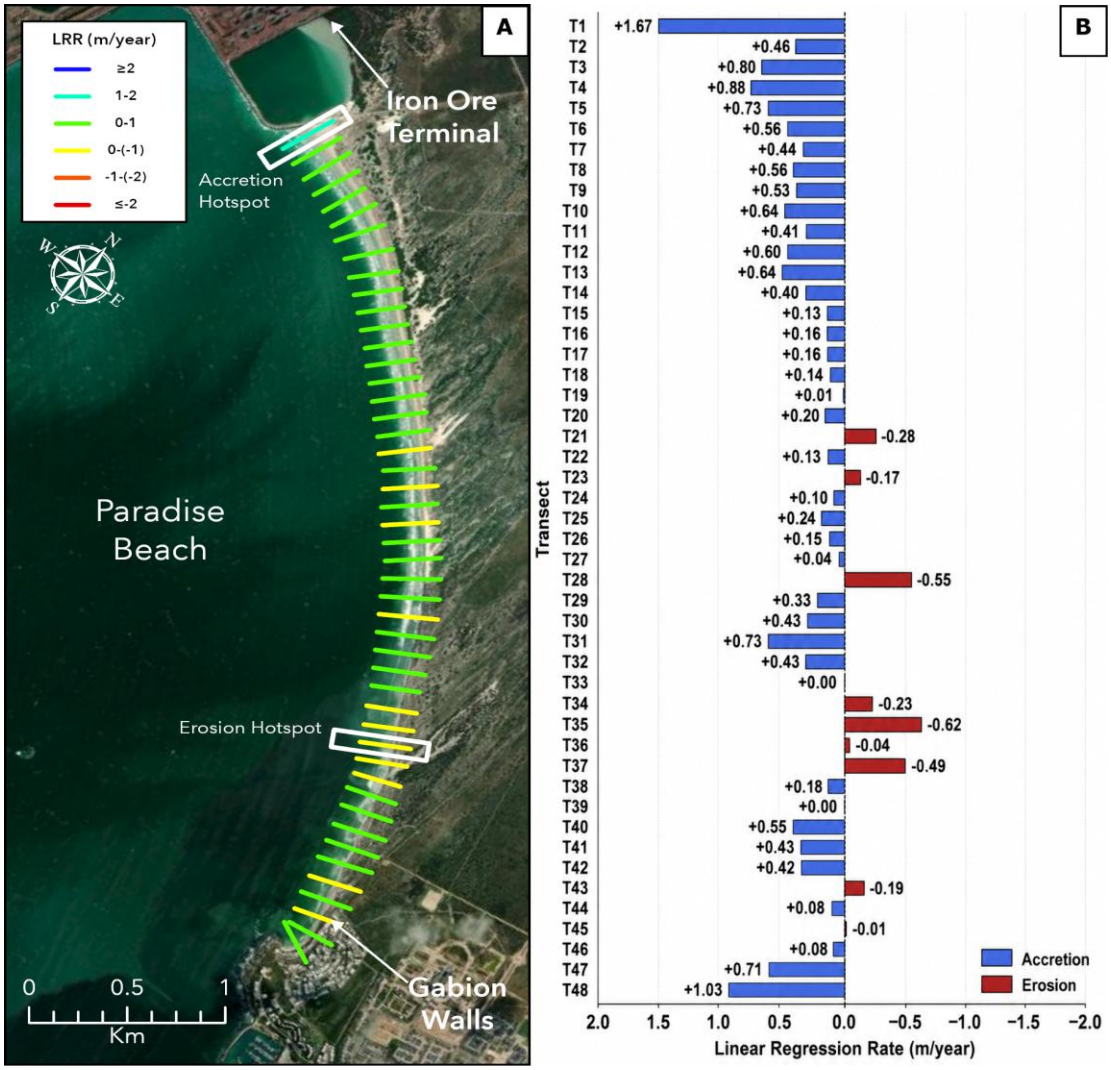


Figure 7.7. 5-year Linear Regression Rates (A) applied to Paradise Beach, and (B) from baseline (0 m/year) at every 100 m transect along the coast.

At the accretion hotspot, a long-term pattern of shoreline advancement was evident, with an average accumulation rate of approximately 1.6 m/year (Figure 7.7). Summer-to-winter shoreline retreat occurred between 2020 and 2022, with a maximum retreat of approximately 15.4 m in 2020 and a minimum retreat of approximately 0.3 m in 2022. From 2023 onward, however, the seasonal pattern shifted towards summer-to-winter accretion, reaching approximately 19.8 m in 2026. Overall, the shoreline advanced by an average of approximately 3.1 m between summer and winter over the 2020–2026 assessment period.

At the erosion hotspot, the shoreline retreated at an average rate of approximately 0.62 m/year between 2020 and 2026 (Figure 7.8). An accumulation generally occurs during spring and summer, followed by shoreline retreat during winter and spring. Overall, the shoreline retreated by an average of approximately 8.3 m between summer and winter. The greatest seasonal retreat was approximately 14.4 m in 2023 with the greatest accretion being approximately 0.7 m in 2025 (Figure 7.8).

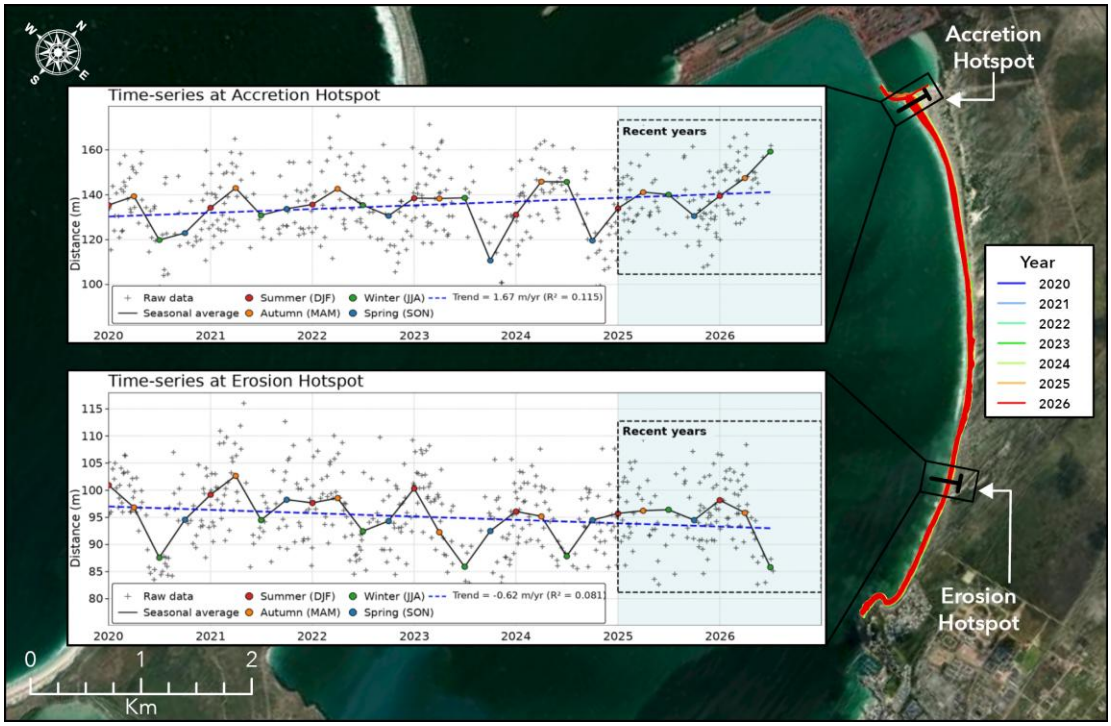


Figure 7.8. Seasonal shoreline changes at Paradise beach (2020–2026), with transects shown in black, baselines as grey dotted lines and shoreline positions color-coded by year. Time-series plots highlight trends at two key locations, showing raw shoreline positions (+), seasonal averages (Summer, Autumn, Winter, Spring) and linear regression lines (dashed blue).

7.3.2 Wave–Shoreline Correlation

Wave correlation analysis along Langebaan Main Beach revealed contrasting patterns between its northern and southern sections (Figure 7.9). Langebaan North Beach, extending from the groynes to the end of the rock revetment, exhibited moderate negative correlations ($r = -0.4$ to -0.6), with higher SWH associated with shoreline erosion. In contrast, Langebaan South Beach, extending from the groynes into the lagoon, displayed moderate positive correlations, with increases in SWH coinciding with seaward shoreline movement.

At Mykonos, Calypso and Leentjiesklip Beaches, correlations ranged from weak to moderate in both directions. Negative correlations occurred more frequently in the northern sections, whereas statistically significant transects in the southern sections generally displayed positive correlations. The remaining transects were not statistically significant ($p \geq 0.05$), demonstrating that the relationship was not consistent across the entire shoreline.

Incoming waves predominantly approached from the south-west, with seasonal erosion most frequently associated with waves from this direction (Figure 7.9).

Wave correlation analysis along Paradise Beach indicated that much of the central section of the beach between the IOT and the gabion walls exhibited a moderate (r = −0.40 to −0.69) to strong (r = −0.70 to −0.89) correlation between SWH and shoreline position (Figure 7.10). This indicates that increases in SWH were generally associated with shoreline erosion along most of the beach. Moderate inverse correlations were observed toward both the northern (near IOT) and southern (near gabion wall) sections of the beach, although these relationships were more widespread in the north than in the south.

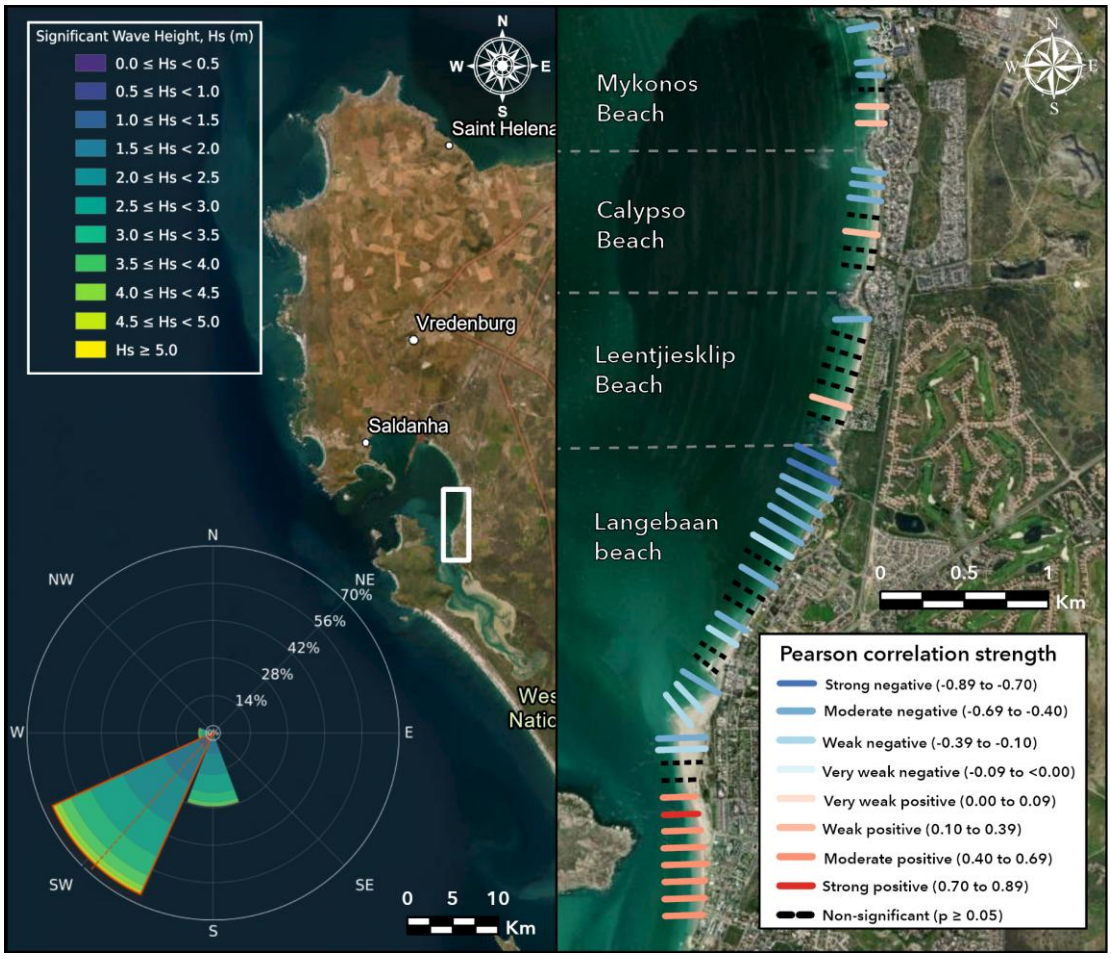


Figure 7.9. Spatial relationship between seasonal changes in SWH and shoreline position along the Langebaan coastline. The left panel shows the study area location, outlined in white, and the distribution and direction of wave heights most frequently associated with erosion. The right panel shows the strength and direction of the Pearson correlations for transects along Mykonos, Calypso, Leentjiesklip and Langebaan Beaches.

In contrast, the northernmost transects, closest to the IOT, and the southernmost transects, near the gabion wall, were not statistically significant ($p > 0.05$). Therefore, there was insufficient evidence to conclude that seasonal changes in wave height were linearly related to shoreline movement at either end of Paradise Beach (Figure 7.10).

Incoming waves along Paradise Beach were found to approach predominantly from the south-west, with waves from this direction most frequently associated with periods of seasonal shoreline erosion (Figure 7.10).

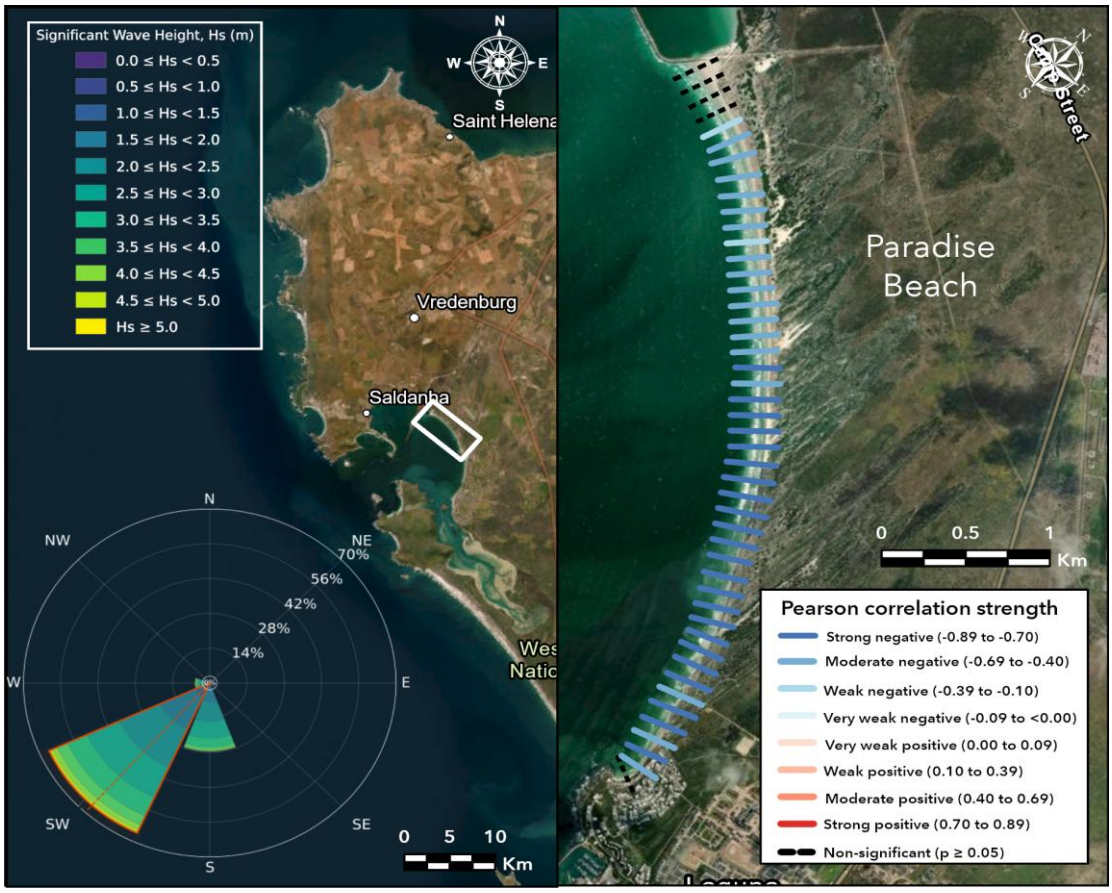


Figure 7.10. Spatial relationship between seasonal changes in SWH and shoreline position along the Paradise coastline. The left panel shows the study area location, outlined in white, and the distribution and direction of wave heights most frequently associated with erosion. The right panel shows the strength and direction of the Pearson correlations for transects along Paradise Beach.

7.4 SUMMARY AND DISCUSSION

Previous efforts presented in the 2025 report used a modified CVI to identify spatial differences in shoreline vulnerability and highlight areas of concern along the Saldanha Bay coastline. While this improved understanding of where erosion and accretion are occurring, it provided limited insight into the physical processes driving these changes. Building on this work, the current monitoring efforts continued the coastal assessment by examining shoreline variability at the seasonal scale over the past five years. This approach was used to determine whether recent shoreline changes represent short-term fluctuations between seasons, or if they are representative of emerging long-term trends at annual or decadal timescales. In addition, SWH was investigated as a potential key driver to determine whether these seasonal shoreline patterns were associated with changes in hydrodynamic conditions.

7.4.1 Sub Annual Coastal Monitoring

The continuation of the coastal monitoring programme demonstrated the value of assessing shoreline change at finer spatial and temporal scales. Most transects along Leentjiesklip, Calypso, and Mykonos beaches showed moderate to low vulnerability, with shoreline changes ranging from relatively stable conditions to sediment build-up of approximately 0–1 m/year over the last five years. However, more pronounced and contrasting changes were recorded around the groyne structures on Langebaan Main Beach.

Greater sediment accumulation was recorded south of the groyne structures opposite Schaapen Island on Langebaan South Beach. Over the last five years, the shoreline in this area has advanced significantly over the last decade and at an average rate of approximately 15 m/year over just the last five years, indicating very low vulnerability to erosion compared with the surrounding coastline.

Although the longer-term record shows that the shoreline has continued to advance as sediment has accumulated, this overall trend has tended to mask smaller seasonal changes. The seasonal assessment nevertheless identified short periods of erosion, mainly during winter and spring, followed by shoreline recovery and further sediment build-up during summer. These seasonal changes have become more noticeable since 2023, whereas earlier years showed relatively little difference between seasons. This emerging pattern may suggest that the longer-term build-up of sediment is beginning to slow, making the natural seasonal cycle of erosion and recovery more visible. The erosion recorded during winter, and spring may also be related to stronger wave conditions and individual storm events, which are examined further in the wave-dynamics assessment (see below).

In contrast, the shoreline directly between the groynes, near Noord Street, remained an erosion hotspot. This area has continued to retreat over the past decade, with an average erosion rate of approximately 2.3 m/year over the last five years. The seasonal analysis showed that most of this erosion occurred during winter, when the shoreline retreated by an average of almost 12 m between the summer and winter season. Although some recovery occurred during the calmer seasons, it was not sufficient to offset the longer-term erosion trend. This indicates that the area is particularly vulnerable during winter, when stronger waves and individual storm events may cause substantial shoreline retreat.

Together, these shoreline responses indicate that, although the groynes have helped slow the prevailing erosion in some adjacent areas, they have not fully stabilised the surrounding coastline along Langebaan Main Beach. This pattern is consistent with the way groynes influence longshore sediment transport. Groyne structures are built to slow the natural movement of sand along the coast by blocking sediment on the updrift side (the side where waves and currents bring in sediment) (Van Rijn 2011). When two groynes are placed some distance apart, they can trap sand on both sides, which helps build up the beach in those areas as observed with accretion present adjacent to the groyne structure. However, this creates a problem for the area in between the groynes, as sand is trapped on either end of the groynes, less sediment reaches in between, causing it to erode more rapidly. This process, known as downdrift erosion, occurs because the natural supply of lateral sand movement is cut off (Bruun 1995). Without enough

sediment arriving to replace what is lost, the beach continues to shrink, increasing the vulnerability of the coastline to erosion in that zone (Figure 7.11).

The ongoing loss of sediment from the coastline between the groynes is increasing the vulnerability of both the natural coastal environment and nearby built infrastructure. As the beach narrows, the natural buffer between the ocean and human development is reduced, leaving beachfront properties, roads, and other critical infrastructure more exposed to coastal hazards. Storm surges, flooding, and energetic wave conditions can further exacerbate shoreline retreat and increase the potential for damage, particularly during extreme weather events (Harley et al. 2022). The severity of this risk is demonstrated by historical erosion events during the 1990s, when two beachfront houses opposite the Langebaan town centre collapsed and were lost. Although groynes and rock revetments were subsequently constructed to stabilise the shoreline, the continued erosion between the groynes indicates that while these measures have reduced vulnerability in some areas they have not been found to eliminate risk throughout the wider coastal region.

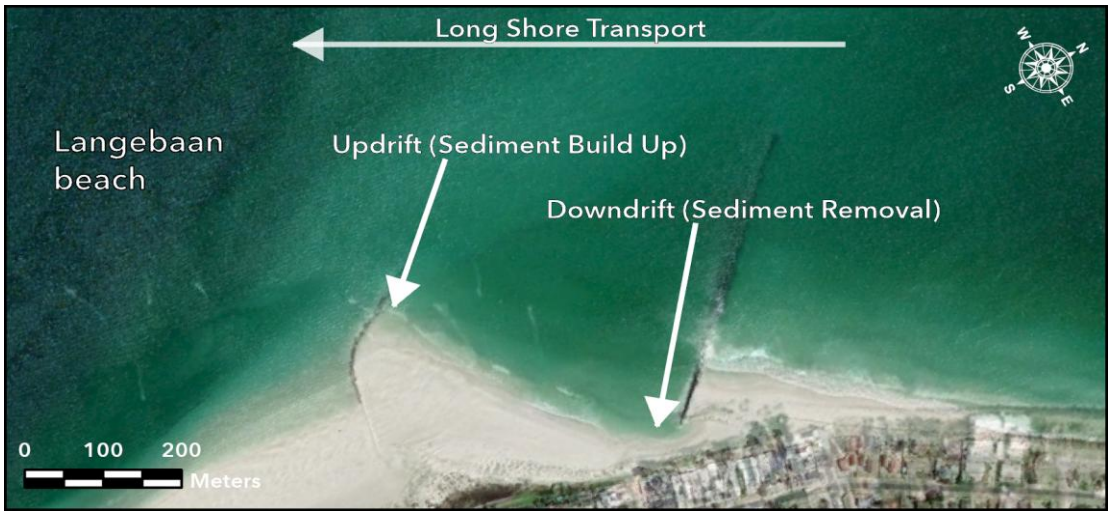


Figure 7.11. The longshore transport of sediment is associated with multiple groyne structures positioned close together. Updrift (sediment accretion) happens on the upstream side of the groyne while downdrift (sediment erosion) happens on the downstream side of the groyne.

If the current linear rate of beach loss continues, which is estimated to be approximately 2.3 m per year, the section of coast located between the Langebaan groynes is projected to experience an additional 11.5 m of shoreline erosion over the next five years. Such a trend could lead to significant land loss to the same extent or even worse that previously seen in the 1990s, increasing the risk of property damage or loss, and potentially disrupt local ecosystems and tourism activities. This ongoing degradation calls for urgent coastal management intervention to mitigate further erosion, protect infrastructure, and preserve the natural coastal environment.

Within Paradise Beach although most of the beach is found to exhibits moderate coastal vulnerability overall ranging from 0 to 1 m/year, the coastal analysis identified several sections within the central part of the beach that continue to experience persistent, low-rate erosion. Shoreline retreat in these areas ranged from approximately 0.04 to 0.62 m/year. Historically, substantially greater rates of erosion were recorded along

parts of Paradise Beach between 2000 and 2005; however, additional monitoring indicates that the rate of retreat has generally slowed and become more stable over the past several years (Clark et al. 2024, 2025).

Although this longer-term trend suggests increasing shoreline stability, the finer-scale seasonal assessment along the hotspot areas showed that the beach continues to undergo periods of erosion and recovery throughout the year. Shoreline retreat was generally more apparent during the higher-energy winter and spring months, while shoreline recovery and sediment build-up were more commonly observed during the calmer summer months in both areas. This shows that, despite relatively low annual erosion rates, individual seasons can still produce noticeable changes in shoreline position that are not fully represented by the longer-term average rate of change.

The continued concentration of erosion within the central section of Paradise Beach suggests that this area may be more sensitive to changing coastal conditions than sections closer to the IOT or gabion walls, however. Additional factors such as stronger wave conditions and individual storm events may therefore contribute to periods of more rapid shoreline retreat in this central section. Repeated seasonal erosion remains important because, where recovery is incomplete, these short-term losses can gradually contribute to longer-term beach narrowing.

Therefore, relative stability in shoreline position does not necessarily indicate stability of the wider coastal system. Along much of Paradise Beach specifically within the central sections, the landward edge of the beach is underlain by exposed calcrete, an indurated calcium-carbonate-rich material that forms a relatively resistant substrate at the base of the coastal dunes (Figure 7.11). The geological development of calcareous and aeolian deposits is well documented within the Saldanha Bay region (Franceschini 2003). While these resistant outcrops are meant to stabilise and locally restrict further landward movement of the shoreline, in-situ observations along Paradise Beach indicate that erosion of the overlying dune and beach have progressively exposed and, in places, facilitated erosion of the underlying calcrete (Figure 7.12).

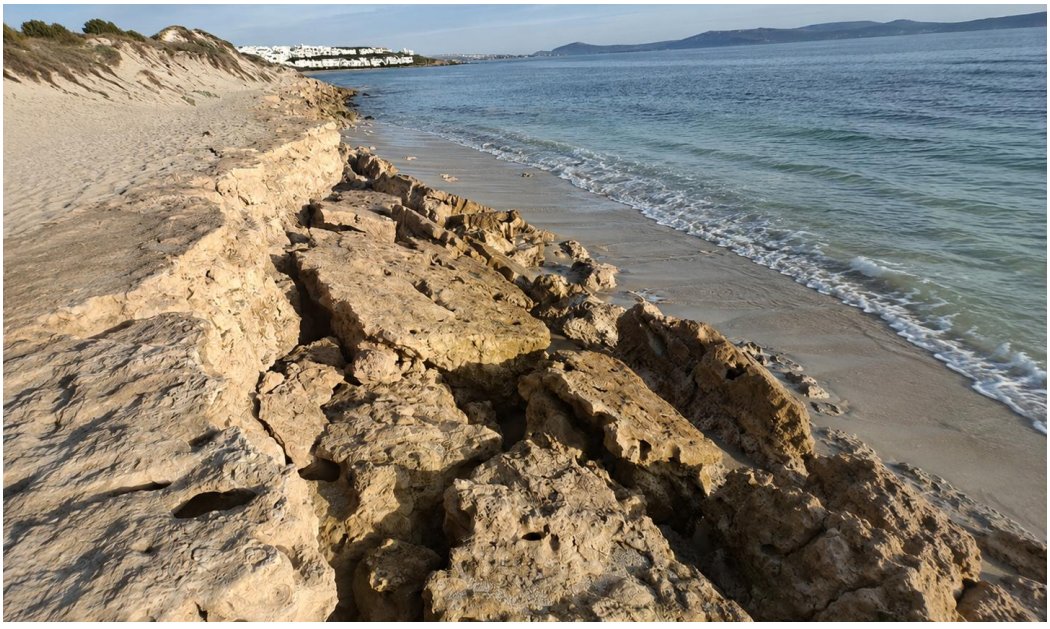


Figure 7.12 Exposed calcrete outcrops along Paradise Beach (Photo: Zac Christensen).

The deterioration of the dune, beach and exposed calcrete therefore represents an additional component of coastal vulnerability that is not fully captured by shoreline-position analysis alone. Continued degradation of these interconnected natural barriers could progressively weaken their ability to dissipate wave energy and protect the landward coastline, increasing its susceptibility to erosion during future storms and other high-energy wave events. Consequently, assessments of coastal stability along Paradise Beach should consider not only changes in shoreline position, but also the physical condition of the beach, dunes and exposed geological substrate through additional monitoring methods.

7.4.2 Correlation of Hydrodynamic drivers

While the continued coastal monitoring program provided valuable insight into the current seasonal patterns of shoreline change, the incorporation of hydrodynamic factors was an important component in developing a more comprehensive understanding of the processes driving coastal dynamics. Examining seasonal patterns of erosion and accretion alongside hydrodynamic conditions has been found to provide important insight into how shoreline change responds to seasonally varying natural forces, including winds and waves (Slott et al. 2006, Murray et al. 2023, Warrick et al. 2025). This information is particularly valuable when planning future erosion control measures in addition to those already in place, as coastal vulnerability the wave dynamics as well as other coastal variables can affect how effective these measures might be (Griggs 2024).

The incorporation of hydrodynamic variables into the 2026 assessment provided further insight into these spatially variable shoreline responses. Along the northern section of Langebaan Main Beach, extending from the groynes through the rock-revetments, significant moderate inverse correlations were identified between SWH and shoreline position. The strongest inverse relationship occurred within the downdrift area between the groynes, which was also identified as the erosion hotspot. In this area, higher winter wave heights were generally associated with landward shoreline movement and erosion, whereas lower summer wave heights were associated with shoreline recovery and accretion. This spatial pattern is consistent with the established understanding that dredging of the historic ebb-tide delta at the lagoon mouth reduced natural wave-energy dissipation. The dredging increased incident wave heights (and thus wave energy) by about 50% along Langebaan beach, with the majority of this added energy focused on the northern shoreline (Flemming 2016)

In contrast, the southern section of Langebaan Beach generally showed the opposite seasonal pattern. The shoreline tended to advance during winter, when offshore waves were larger, and retreat during summer, when wave heights decreased. This pattern is consistent with earlier monitoring by the WQT, which recorded summer erosion south of the groynes and winter erosion north of the structures (Clark et al. 2024)(Section 7.1.1). However, this indicates that larger offshore waves are not directly responsible for erosion along Langebaan South Beach. Instead, winter accretion along the beach may result from the redistribution of material from Langebaan North Beach. During summer, local processes within the sheltered lagoon, including tidal currents, locally generated wind waves, sediment transport and human activities, may contribute to shoreline erosion. Future monitoring should investigate these processes in greater detail to improve

understanding of the different factors influencing the northern and southern sections of Langebaan Beach.

A similar northern and southern pattern was observed along Mykonos, Leentjiesklip and Calypso beaches, although most transects were found to have statistically significant relationships. Of the ones that correlated, the northern sections generally experienced erosion as offshore wave heights increased, while the southern sections tended to experience sediment build up under the same conditions. One possible explanation is that incoming waves are more directly focused on the northern sections of these beaches. Higher energy winter waves may therefore erode sediment from these more exposed areas, with some of this material subsequently transported southwards towards the more sheltered sections. During summer, when offshore wave heights decrease, local processes may become more influential and contribute to erosion along parts of the southern shoreline. However, as most of these transects were weak in correlation and therefore a higher temporal resolution analysis is required to fully conclude if this pattern is how changes in wave conditions affect shoreline movement along these coastlines.

Along Paradise Beach, the central shoreline exhibited a strong inverse correlation ($r = -0.70$ to -0.80) coming from the southwest, confirming that increased seasonal wave heights were associated with shoreline retreat. This supports previous observations that wave exposure has progressively eroded the beach and dune system and exposed the underlying calcrete formations (Clark et al. 2024). However, correlations weakened towards the IOT and gabion walls. This may reflect local changes in shoreline orientation, together with sheltering and wave redirection caused by port infrastructure and rock formations. Because shoreline response is influenced by both wave height and wave-approach angle, these results demonstrate that offshore SWH alone may not fully represent the conditions experienced along different parts of the beach and that additional drivers should be considered to fully understand the dynamics of the coastline (Harley et al. 2015, Nobre Silva et al. 2024).

The absence of a significant seasonal correlation seen over certain transects does not mean that a shoreline is unaffected by waves or erosion. Seasonal averaging can mask short-duration storms capable of causing rapid and substantial erosion. Event-based monitoring can therefore provide additional information on the immediate shoreline response, subsequent recovery, and areas that are particularly vulnerable to extreme condition (Nguyen et al. 2016, Leach et al. 2023, Murray et al. 2023, Clark et al. 2024). An example of the information obtainable at this finer temporal scale was done on an extreme storm event seen on 7 June 2017, when SWH reached 6.32 m. Satellite images acquired before and after the event showed substantial shoreline movement across most transects between Langebaan Main Beach and Mykonos. An individual investigation was done at the erosion hotspot between the groynes showing the extracted shoreline positions differed by approximately 20 m with observed recovery starting 2–3 days after the event.

However, cloud cover prevented direct observation during the storm, with the nearest usable images acquired on 31 May and 20 June 2017. The observed shoreline change therefore cannot be attributed entirely to the storm, as other processes may have influenced the beach between these dates. Nevertheless, the assessment indicates that short-duration wave events may cause substantial shoreline change that is not captured by seasonal wave correlations. Higher-temporal-resolution monitoring is therefore

needed to better determine how individual changes in wave conditions influence shoreline erosion and recovery.

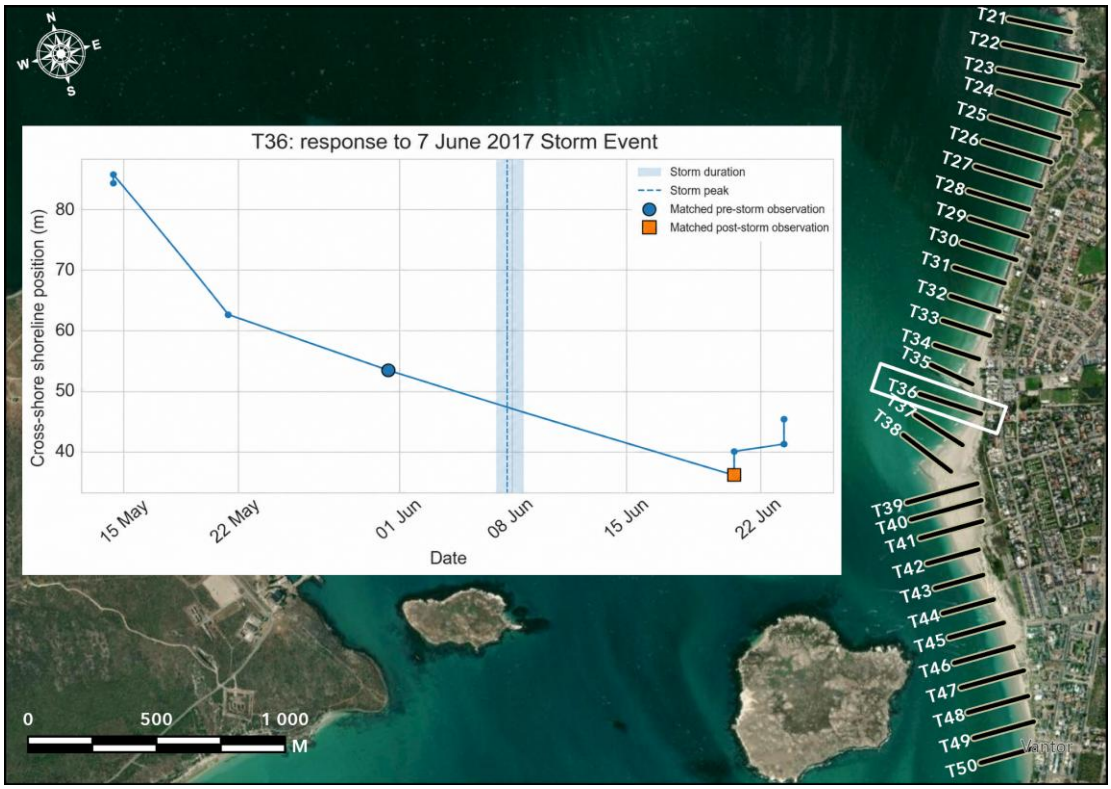


Figure 7.13. Location of Transect 36 and changes in shoreline position before and after the 7 June 2017 storm event, including the closest available pre- and post-storm observations.

Aerial view of erosion damage at Paradise Beach

8 BENTHIC SYSTEMS

8.1 INTRODUCTION

Monitoring the biological components of an ecosystem, in addition to physico-chemical and eco-toxicological variables, is critical as biological indicators provide a direct measure of the ecosystem's state at a selected point in space and time. Among these indicators, benthic macrofauna are the most frequently used to detect changes in the health of the marine environment. This preference stems from their short lifespan, causing their community composition to respond rapidly to environmental changes (Warwick 1993). Given their relative immobility, compared with fish and birds, they tend to be directly affected by pollution and are easy to sample quantitatively (Warwick 1993). In addition, benthic macrofauna are well-studied compared to other sediment-dwelling fauna (e.g. meiofauna and microfauna), and taxonomic keys are available for most groups. Furthermore, their community responses to various anthropogenic influences are well-documented.

Benthic systems include both infauna and epifauna as key components of their ecosystems. Infauna are organisms that live within the sediments of the seafloor, such as burrowing clams, worms, and polychaetes. They often dominate soft sediment communities and play essential roles in nutrient cycling and organic matter breakdown by burrowing and bioturbation. Epifauna, on the other hand, live on the surface of the seafloor or attached to substrata. Examples include ascidians, mussels, barnacles, starfish, and crabs. They are especially abundant in rocky or reef environments and contribute to the ecosystem as predators, grazers, and suspension feeders.

Organic matter is one of the most universal pollutants affecting marine life and can lead to significant changes in community composition, particularly in semi-enclosed or closed bays with restricted water circulation, such as Saldanha Bay. High organic loading often leads to eutrophication, resulting in various responses to benthic infaunal community structure, for example: increased growth rates, species disappearance, and a reduction in species numbers. In severely eutrophic and anoxic sediments, benthic organisms may even disappear entirely (Warwick 1993). Additionally, the community composition of benthic infauna is likely to be impacted by increased levels of other contaminants, such as trace metals and hydrocarbons in sediments. Areas frequently disturbed by mechanical activities (like dredging), tend to be dominated by opportunistic pioneer species, rather than longer-lived species.

The main aim of monitoring the health of an area is to detect the effects of stress and to monitor recovery following environmental perturbation. Various indices based on benthic invertebrate fauna data can be used to reveal conditions and trends in ecosystem health. These indices include measures of community composition, diversity and species abundance, and biomass. Given the complexity of environmental assessment, it is recommended to use multiple indices (Salas et al. 2006). This report considers the community composition, diversity, abundance and biomass of soft bottom benthic infauna samples collected in Saldanha Bay and Langebaan Lagoon from 1999 to 2026.

Traditionally, sediment quality results were presented as a stand-alone chapter in this report. However, due to budgetary constraints, it has been agreed that sediment quality

monitoring will now occur biennially rather than annually. To aid in the interpretation of benthic macrofauna results, a summary of historical sediment quality data from the five most recent sampling events has been included to provide context for the interpretation of the benthic macrofauna monitoring results.

8.2 RECENT AND HISTORIC DATA ON BENTHIC MACROFAUNA COMMUNITIES

8.2.1 BENTHIC INFAUNA

Historical surveys of benthic infauna in Saldanha Bay date back to the 1940s but are not comparable to the present survey due to differing methodologies. For example, early studies, like Moldan (1978) and Christie & Moldan (1977) used different sampling equipment and sampling techniques, complicating direct comparisons. However, more recent studies by the CSIR (Bickerton 1999) and Anchor Environmental Consultants (2004–2026) were consistent in their sampling techniques. Consequently, this report focuses on data from 1999 to 2026. It was found, however, that biomass data collected from sites in Langebaan Lagoon in 1975, are directly comparable with recent data and have been included in analyses where possible.

Benthic infauna have been sampled at over 30 sites in Big Bay (nine sites), Small Bay (ten sites) and Langebaan Lagoon (12 sites) since the State of the Bay (SOB) monitoring programme began in 2004. In 2021, 21 additional sites were sampled as part of the Aquaculture Development Zone (ADZ) Monitoring Survey in Saldanha, including 11 impact sites within the Big Bay ADZ (three within the fin fish area and eight in the shellfish area), four impact sites within the Outer Bay North ADZ, and six reference sites outside the ADZ boundaries in Big Bay (n=3) and Outer Bay North (n=3). The study aimed to determine whether the aquaculture operations were causing measurable benthic impacts (Probyn et al. 2023). The sediment thickness was less than 20 cm, at most ADZ sites, thereby excluding large burrowing taxa (such as prawns) and explaining the lower biomass observed. However, the abundance of benthic infauna observed at ADZ sites was comparable to that at traditional SOB monitoring sites due to the presence of numerous smaller surface-dwelling taxa (amphipods, isopods and various polychaetes), typically found in the top sediment layer.

In 2016, the SOB monitoring activities were expanded to include monitoring of benthic infauna at three new sampling sites near the head of the Lagoon at Elandsfontein. Concern had been raised around potential impacts that the proposed phosphate mine at Elandsfontein might have on groundwater quality and flows into Langebaan Lagoon; hence the objective to establish an appropriate baseline of the present benthic infauna community structure against which any potential future changes in the Lagoon can be benchmarked. The eighth set of baseline results are presented here and are assessed in context of the entire Saldanha Bay/Langebaan Lagoon system.

In the 2025 survey, the number of sampling sites in the Bay Area and Lagoon was reduced to nearly half compared to 2024 due to budget constraints. Specifically: Small Bay decreased from ten sites in 2024 to six sites in 2025; Big Bay from nine sites to six; and Langebaan Lagoon from nine sites to four. In 2026, sampling returned to the full suite of sites across the Bay Area and Lagoon.

8.2.2 BENTHIC EPIFAUNA

The Saldanha Bay sea-based ADZ baseline benthic survey report (Mostert et al. 2020b) mentioned that divers encountered calcrete reef at some sampling sites during the 2019 baseline survey (Capfish *pers. comm.* 2019) and had difficulty in obtaining grab samples in Big Bay during the 2020 ADZ physicochemical sediment surveys (Mostert et al. 2020a). Divers from Anchor Environmental Consultants deploying water quality monitoring instruments in April 2020 confirmed the presence of reef in several areas of the Big Bay ADZ precinct. A literature review further revealed an extensive abrasion platform, consisting of exposed calcrete rock, throughout much of Big Bay (Flemming 1977). Furthermore, the finfish farming lease holder at the time provided a bathymetry map indicating extensive low-profile reef across the area (Mostert et al. 2020b).

Photographs of the benthic environment taken by Anchor Environmental Consultants confirmed the presence of low-lying reef, which was possibly periodically inundated with a fine layer of sediment (Mostert et al. 2020a, b). These exposed reef patches supported upright epifauna such as basket stars, sponges, and bryozoans, and provided habitat for West Coast rock lobster *Jasus lalandii*. Overall, the reef was characterised as low-profile, generally <1 m in height above the seafloor and subject to natural sand inundation. However, substantial outcrops >1 m in height were also observed, supporting a well-established epifaunal community.

During an ADZ monitoring survey conducted in 2022, Anchor Environmental divers collected video footage at two sites where reef was encountered (BB5 and FF2) and a qualitative description of the reef epibenthos at each site was undertaken. A total of 21 species were identified, with more species recorded at site BB5 (20 species) than at the finfish site FF2 (10 species). However, the sites shared several common species including West Coast rock lobster, red starfish *Patiria granifera* and reticulated starfish *Henricia ornata*, Cape sea urchin *Parechinus angulosus*, and beds of the common feather star *Comanthus wahlbergii*. Examples of these are shown in Figure 8.1.

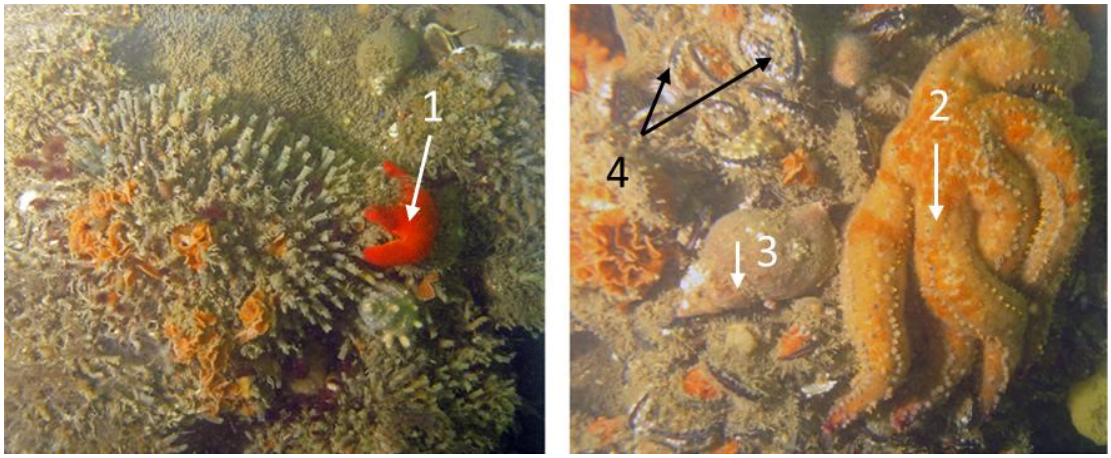


Figure 8.1. Stills taken from video footage of ADZ Big Bay farm site BB5 showing examples of epifaunal reef community. 1) *Patiria granifera*, 2) *Marthasterias africana*, 3) *Burnupena* sp., 4) *Aulacomya ater*, 5) *Comanthus wahlbergii*, 6) *Jasus lalandii*, 7) *Parechinus angulosus*.

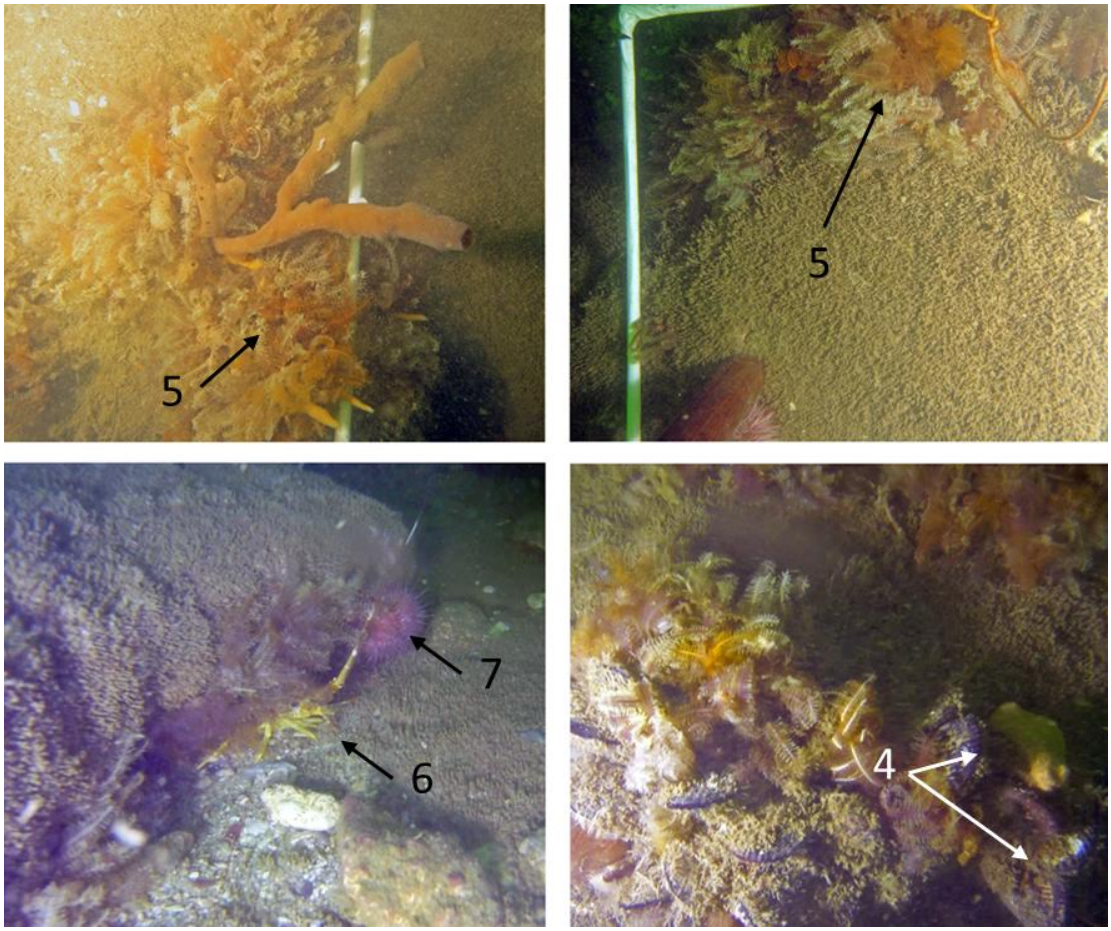


Figure 8.1 continued. Stills taken from video footage of ADZ Big Bay farm site BB5 showing examples of epifaunal reef community. (1) *Patiria granifera*, (2) *Marthasterias africana*, (3) *Burnupena* sp., (4) *Aulacomya ater*, (5) *Comanthus wahlbergii*, (6) *Jasus lalandii*, (7) *Parechinus angulosus*.

In 2024, a bathymetric survey was conducted to accurately map the extent and distribution of the Big Bay reef habitat to support ongoing monitoring of potential aquaculture impacts (Hutchings et al. 2024) (Figure 8.2). Additionally, a repeat ecological survey of the reef habitat was carried out in 2025 (Figure 8.2). For this, six previously established reef sites (three impact and three control, including one finfish baseline site) were resampled using standardized diver-based video transects, photo-quadrats, and qualitative specimen collections to assess epifaunal communities. Community composition was compared among impact, control, and baseline sites using presence/absence data and multivariate statistical analyses.

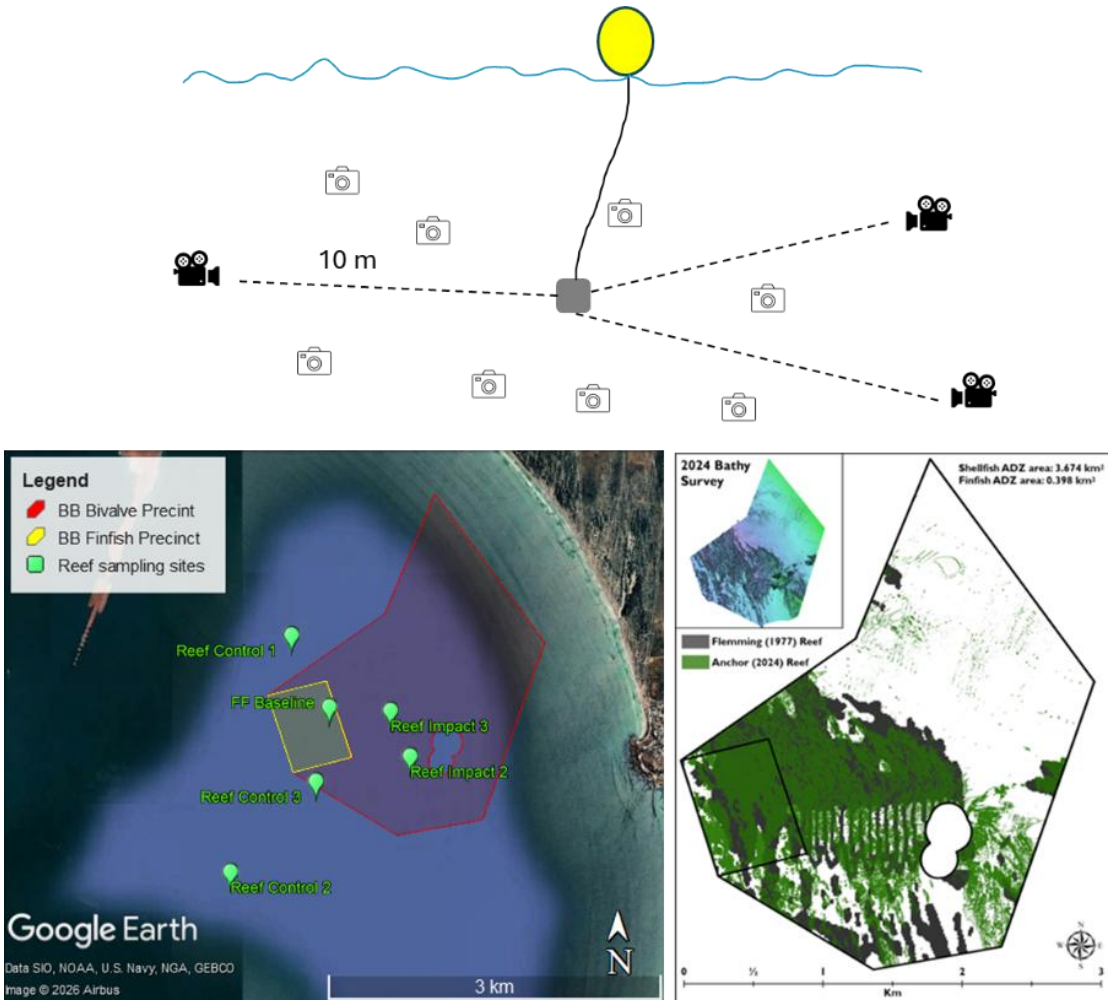


Figure 8.2. Location of the reef survey sites in Big Bay in 2025 (Van Rensburg et al. 2025). Diagram of the survey method (Top) and location of the reef survey sites in Big Bay (bottom left) employed by Dawson et al. (2022) and replicated in this study, and extent of the Big Bay precinct calcrete abrasion reef as updated in 2024 (Hutchings et al. 2024) (bottom right).

Results showed that epifaunal communities were broadly similar across impact, control, and baseline sites (Figure 8.3). While a statistically significant difference was detected between impact and control sites, site category explained only 2.6% of the variation in community composition compared with 19.3% explained by differences among individual sampling sites (Van Rensburg et al. 2025). Consequently, no clear effects attributable to current aquaculture activities could be identified. Bathymetric mapping showed that much of the reef occurs within the ADZ, particularly the finfish precinct, leading to a recommendation that future finfish aquaculture be directed to the Outer Bay to reduce potential impacts on reef habitat. Continued monitoring was also recommended given the ecological importance of the reef, including its role as a nursery habitat for juvenile West Coast rock lobster.

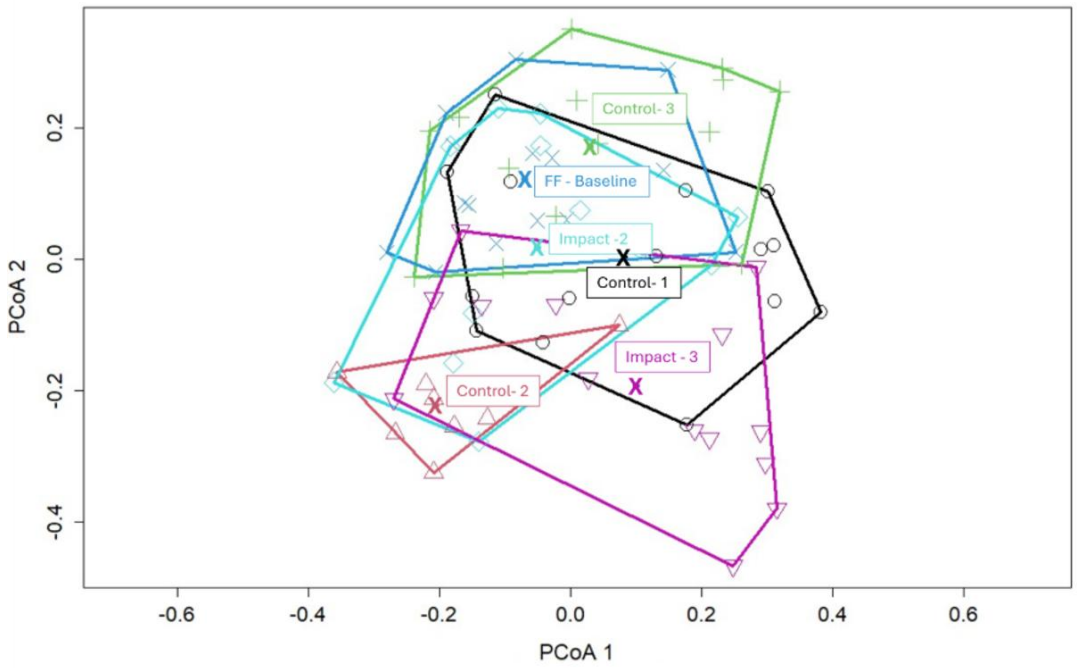


Figure 8.3. Dispersion plot comparing macrofauna species richness between sites within the control and impact reef survey treatments. Dark solid crosses indicate site centroids 2025 (Van Rensburg et al. 2025).

8.3 APPROACH AND METHODS

8.3.1 SEDIMENT SAMPLING AND ANALYSIS

Sediment samples were collected from a total of 31 sites in Saldanha Bay and Langebaan Lagoon in March/April 2020, 2021, 2022, 2023, and 2025, as part of the annual SOB sampling programme (Figure 8.4). These samples were analysed by accredited laboratories for their sediment particle size distribution, organic carbon and nitrogen content, and trace metal content using a suite of methodologies.

To aid understanding of the toxicity levels of trace metals in the sediment, and how they may affect biota, trace metal values were compared to the National Oceanic and Atmospheric Administration (NOAA) Effects Range Low (ERL) concentration thresholds (Table 8.1) (Long et al. 1995, Buchman 2008). The ERL represents the concentration at which toxicity may begin to be observed in sensitive species and is calculated as the lower 10th percentile of sediment concentrations reported in literature that co-occur with any biological effect.

However, high metal concentrations in sediment may simply reflect the natural mineralogical composition of the parent material, granulometry and organic matter content of the host sediment, and do not automatically indicate that contamination has occurred (Newell et al. 1998). Meaningful interpretation of sediment trace metal concentrations must account for naturally occurring baseline concentrations in order to differentiate anthropologically introduced concentrations. Anomalous metal concentrations can be identified through defining the relationship between a metal and a normaliser such as aluminium or iron (Newell et al. 1998).

Table 8.1. The NOAA Effects Range Low (ERL) for trace metals in sediment (Long et al. 1995, Buchman 2008).

Metal	ERL (mg/kg)
Arsenic (As)	8.2
Cadmium (Cd)	1.2
Copper (Cu)	34.0
Chromium (Cr)	81
Lead (Pb)	46.7
Nickel (Ni)	20.9
Zinc (Zn)	150.0

In this case, an Enrichment Factor (EF) was calculated by comparing the concentrations of a given metal of concern and the concentrations of aluminium in a modern sample, with the concentrations of the metal of concern and aluminium in a historical sample. This provides a ratio which can be used to determine the extent of contamination in the sample (Table 8.2). In the absence of appropriate historical values for Saldanha Bay, the background ratios in the Earth's crust were selected for EF analysis, which are sufficient to indicate the approximate degree and distribution of trace metal enrichment within the Bay (CSIR 1999). Relevant crustal values were available for cadmium, copper, nickel, lead, and zinc, so enrichment could only be determined for these five metals. Enrichment Factors ≤1 indicate no elevation relative to pre-development conditions, while Enrichment Factors >1 indicate a degree of metal enrichment from non-crustal materials within the sediments, indicating trace metal origins such as pollution or biota (Barbieri 2016).

Table 8.2. Enrichment Factor (EF) Categories (Barbieri 2016).

EF ratio	Soil dust quality
EF<2	Deficiency to minimal enrichment
2< EF <5	Moderate enrichment
5< EF <20	Significant enrichment
20< EF <40	Very high enrichment
EF > 40	Extremely high enrichment

Other interpretation of the sediment quality results involved the determination of the molar ratios of carbon to nitrogen, i.e., the C:N Redfield ratio. This parameter can assist in identifying sources of organic contamination in sediment (Clarence 1934). Organic matter originating from marine algae (phytoplankton) typically has a C:N ratio between 6 and 8, whereas matter originating from terrestrial plant sources generally exceeds this level. Fish factory waste is nitrogen-rich, therefore extremely low C:N ratios would be expected in the vicinity of a fish waste effluent outfall.

The results from the last five sediment sampling events were collated to provide a general indication of the current state of sediments in the Bay. The spatial patterns in these results across the sampled sites, as well as the relationships between parameters, were assessed to aid interpretation of the benthic macrofauna results.

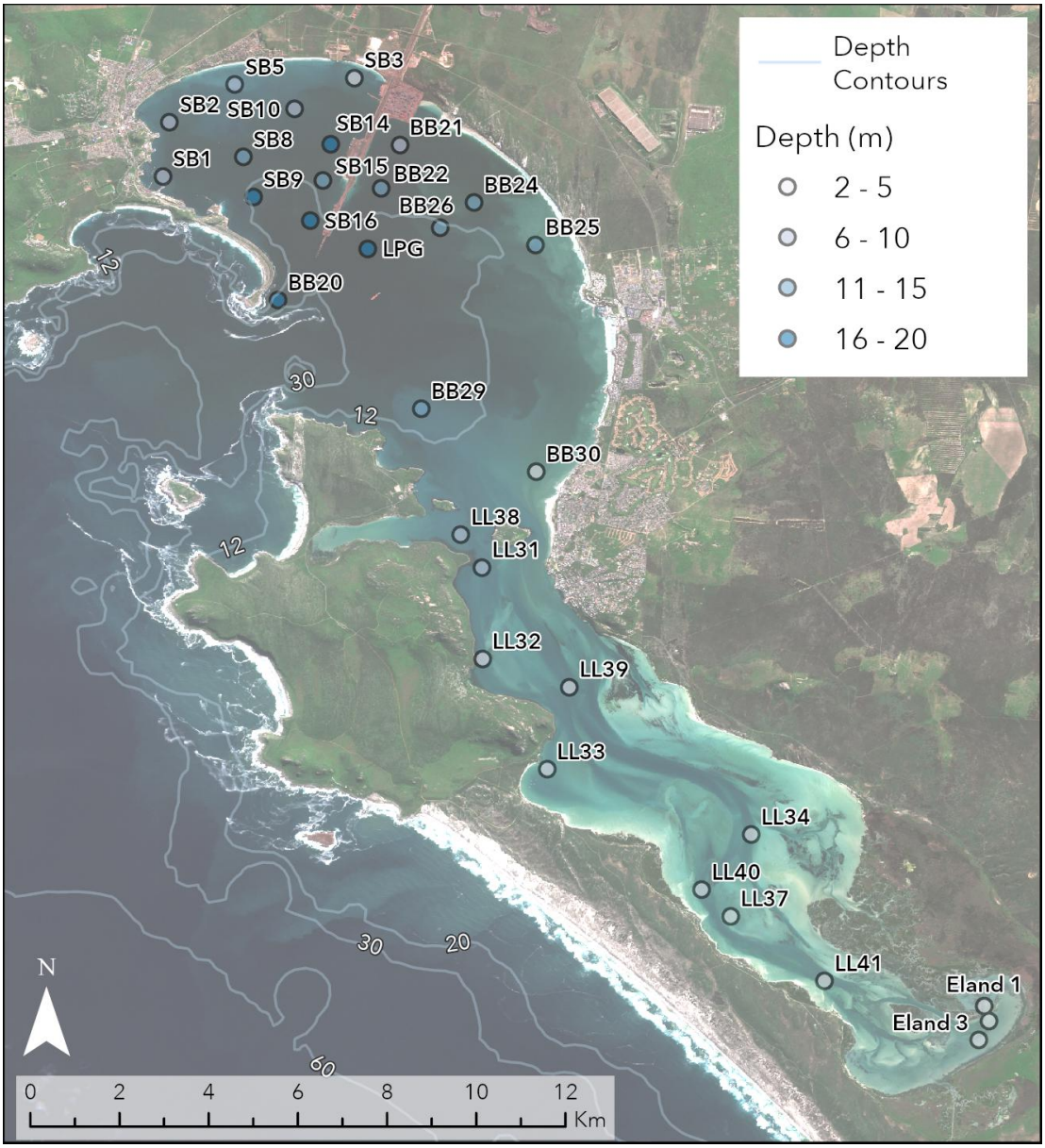


Figure 8.4. Sediment and macrofauna sampling sites and respective depth ranges (m) in Saldanha's Small Bay (SB) and Big Bay (BB), Langebaan Lagoon (LL) and Elandsfontein (Eland).

8.3.2 BENTHIC MACROFAUNA SAMPLING

Benthic macrofauna samples were collected at the same sites as the sediment samples (Figure 8.4) using a diver-operated suction sampler (Figure 8.5A), which sampled an area of 0.12 m^2 to a depth of 30 cm and retained benthic infauna in a 1 mm mesh sieve bag (Figure 8.5B). At sites less than 2 m deep, three hand-core samples were taken, totalling a sampling surface area of 0.08 m^2. In 2016 and 2017, Elandsfontein site samples were collected using the same hand-core method, but from 2018–2023, a Van Veen grab (of 0.14 m^2 bite size) was used. In 2024 to 2026, sample collection reverted to the hand-core method. All infauna abundance and biomass data were standardised per unit area (m^2). Samples were stored in plastic bottles and fixed with 5% formalin (Figure 8.5C).

In the laboratory, samples were rinsed of formalin and all fauna >1 mm in size were removed and preserved in 1% phenoxyethanol (Ethylene glycol monophenyl ether)

solution. Infauna were identified to species level where possible (Figure 8.5D), or at least to family level, and validated using 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.

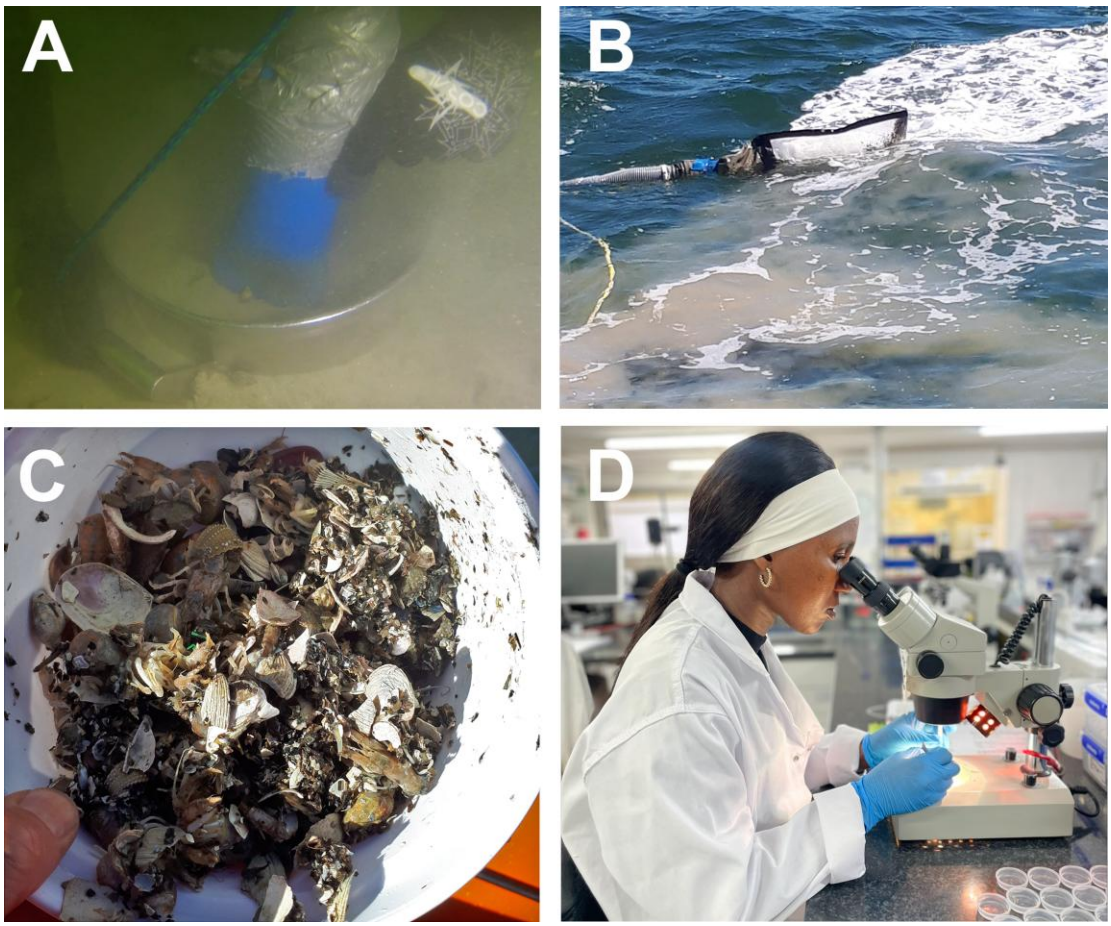


Figure 8.5. (A) benthic macrofauna sample collection with diver-operated suction hose; (B) Sample retained in 1 mm mesh sieve bag; (C) Sieved sample ready for preservation; (D) Identification of benthic macrofauna under stereo dissecting microscope.

8.3.3 BENTHIC MACROFAUNA ANALYSIS

The data collected from this survey served two purposes: 1) to assess spatial variability in the benthic infauna community structure and composition among sites, and 2) to evaluate changes in benthic community structure over time in relation to past surveys. Both spatial and temporal assessments are necessary for providing a comprehensive understanding of the current health of the Bay.

COMMUNITY STRUCTURE AND COMPOSITION

Changes in benthic species composition can be the first indicator of disturbance, as more sensitive species are likely to decrease in abundance under stress, while others are more resilient and may increase, occupying space or resources vacated by the sensitive species. Monitoring temporal variation in community composition provides insights into the rate of ecosystem recovery following disturbances in different areas. This information helps predict the impacts of proposed activities more accurately. “Recovery” following

environmental disturbance is generally defined as the establishment of a successional community of species that progresses towards a community similar in species composition, abundance, and biomass to the one previously present before a disturbance (Newell et al. 1998). The rate of recovery depends on environmental conditions and the communities they support. Given the spatial variation in environmental conditions (largely influenced by depth and exposure) and anthropogenic disturbances throughout Saldanha Bay and Langebaan Lagoon, recovery is known to vary across the system.

Species with high fecundity, rapid growth rates and short life cycles, known as "r-strategist" or opportunistic species, can rapidly invade and colonise disturbed areas (Newell et al. 1998). Their presence generally indicates unpredictable short-term variations in environmental conditions caused by natural factors or anthropogenic activities. In contrast, the community composition in stable environments is predominantly influenced by biological interactions and are dominated by "K-strategists". These species are selected for their competitive ability and are characterised by long lifespans, larger body sizes, delayed reproduction, and low natural mortality rates. Intermediate communities, with varying proportions of opportunistic species and K-strategists, are likely to exist between the extremes of stable and unstable environments.

The statistical programs, R and PRIMER v6, were used to analyse benthic infauna abundance data (Anderson et al. 2008, R Core Team 2022, Posit team 2025). Data were fourth root transformed and converted to a similarity matrix using the Bray-Curtis similarity coefficient. Non-metric Multidimensional Scaling (nMDS) plots were constructed to identify 'natural groupings' between sites for spatial assessment and between years for temporal assessment. The R packages, 'vegan' and 'ggplot2' were used to construct the nMDS plots (Wickham 2008, Oksanen et al. 2024). SIMPER analysis was used to identify species primarily responsible for driving similarity among samples. These multivariate results characterised different regions of the system based on the communities present at the sites. It is important to note that the community composition reflects not only the physicochemical health of the environment but also the ability of communities to recover from disturbance.

Infaunal community descriptors (number of taxa, abundance, evenness, and diversity) were calculated using the DIVERSE function in PRIMER V6 (Clarke & Warwick 2001). Diversity indices provide a measure of how individuals are distributed among different species, offering insight into the overall diversity of a community. Changes in benthic diversity are important to monitor because increasing levels of environmental stress generally decrease diversity. Community diversity is influenced by two main aspects: species richness (the total number of species present) and evenness (how evenly individuals are distributed among species). A community with greater evenness is considered to be more diverse.

The Shannon Diversity Index (H') was calculated for each sampling location using PRIMER V6, with the following formula:

$$H' = -\sum_{i=1}^{s} p_i \ln p_i$$

Where p_i is the proportion of the total count represented by species *i*. This measurement is the most used diversity measure, as it incorporates both species richness and equability.

The H' index considers both species richness and evenness of the species' abundance, providing a single number that reflects the distribution of individuals among species in a community. A higher H' value indicates a more diverse community, with greater evenness and a larger number of species. This index was plotted for plotted geographically for each site and used to interpolate values for the entire system using ArcGIS, revealing spatial patterns. Species richness (also known as alpha diversity) was also calculated for the pre-designated locations from past surveys (1999 to 2026) in Small Bay, Big Bay, Langebaan Lagoon, and Elandsfontein. When interpreting diversity values, it is important to consider that factors such as predation, competition, and disturbance all shape a community. Therefore, physical parameters and other biotic indices should also be considered when drawing conclusions from a diversity index.

The relationship between biotic (macrofaunal abundance) and abiotic variables (sediment grain size composition, depth, sediment grain size fractions, Total Organic Carbon (TOC), Total Organic Nitrogen (TON), C:N, and trace metals) was investigated using a distance-based linear model (DistLM) (Anderson et al. 2008, Oksanen et al. 2024). DistLM is used to identify which subset of environmental variables best explains variation in macrofaunal community composition, by fitting linear models to a dissimilarity matrix (e.g. Bray-Curtis) and selecting parsimonious models via permutation tests and criteria such as AICc or adjusted R^2. The resulting DistLM model is then visualised with distance-based redundancy analysis (dbRDA), a constrained ordination that projects the macrofaunal samples into multivariate space along axes that represent the variation explained by the selected environmental predictors, allowing you to interpret the direction and strength of each variable's effect on community structure (Oksanen et al. 2024).

8.4 SEDIMENT COMPOSITION

8.4.1 BACKGROUND

Sediment quality is an important indicator of ecosystem health as it reflects alterations in benthic sediments from their natural state, through factors like particle size distribution, organic content, and contaminant levels. The proportion of various sediment grain sizes play a pivotal role in shaping biological communities and determining the potential for organic and contaminant accumulation (Clabaut & Davoult 1989, Martinez-Garcia et al. 2015, Kim et al. 2020). 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. Mud (sediment with a grain size <62.5 µm) content is important to monitor as higher proportions of mud, relative to sand or gravel, can lead to high organic loading and trace metal contamination, thereby affecting benthic macrofauna through various responses such as increased growth rates, species loss due to oxygen depletion, and shifts in community composition (Warwick 1993, Hyland et al. 2005, Dewenter et al. 2023). Disturbances such as dredging activities can resuspend mud, releasing pollutants into the water column (Eggleton & Thomas 2004). Although

buried metals are typically inert, they can become hazardous once transformed into soluble forms like metal sulphides.

Trace or heavy metals occur naturally in the marine environment, and while some are known to provide important micronutrients for living organisms (e.g. iron, zinc, manganese, copper, cobalt, molybdenum and nickel), others (e.g. lead, silver and mercury) are biological inhibitors which are not known to assist with any metabolic functions (Sunda 1989, Roesijadi & Robinson 1994). At elevated levels, however, all trace metals and even important micronutrients, can become toxic (Sunda 1989). Disturbance to the environment by either anthropogenic or natural factors can lead to an increase in metal concentrations above established safety thresholds, which can result in negative impacts on marine organisms, especially filter feeders such as mussels that tend to accumulate metals in their flesh (Andersen et al. 1996, Pérez-López et al. 2003). Elevated trace metal concentrations in the marine environment as a result of anthropogenic activity have been shown to decrease aquatic diversity (Andersen et al. 1996).

Total organic carbon (TOC) and total organic nitrogen (TON) accumulate in the same areas as mud, as organic particulate matter is of a similar particle size range and density to that of mud particles and tends to settle out of the water column together with the mud. These tend to be sheltered areas with low current strengths and limited wave action.

The most likely sources of organic matter in Saldanha Bay are from phytoplankton production and its decomposition, 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, and stormwater. The accumulation of organic matter in sediments doesn't necessarily directly impact the environment but the bacterial breakdown of the organic matter can 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 are characteristically black, foul smelling and toxic for living organisms.

8.4.2 RESULTS

The sediment at all sites sampled within Saldanha Bay and Langebaan Lagoon was predominantly sandy, and in the last five sampling events, was composed of an average of 96.0% sand, with small components of gravel and mud (Figure 8.6). In general, Small Bay tended to be slightly more muddy than other areas, with an average of 4.7% mud, followed by Big Bay, which had an average of 3.6% mud. Langebaan Lagoon was more strongly sand-dominated, with an average of 97.8% sand and only 0.7% mud, while Elandsfontein comprised 98.0% sand and 1.9% mud. However, variation among all sites was high. Site SB1 stood out from the other sites with the highest average mud content (15.2%) and lowest sand content (84.6%). Other relatively muddy sites included SB9 (8.2%), SB14 (8.1%), and BB26 (8.4%). In contrast, BB30, LL39, and LL40 all had averages of less than 0.3% mud. LL38 in Langebaan Lagoon contained more gravel than the other sites, comprising an average of 10.8% gravel (Figure 8.6).

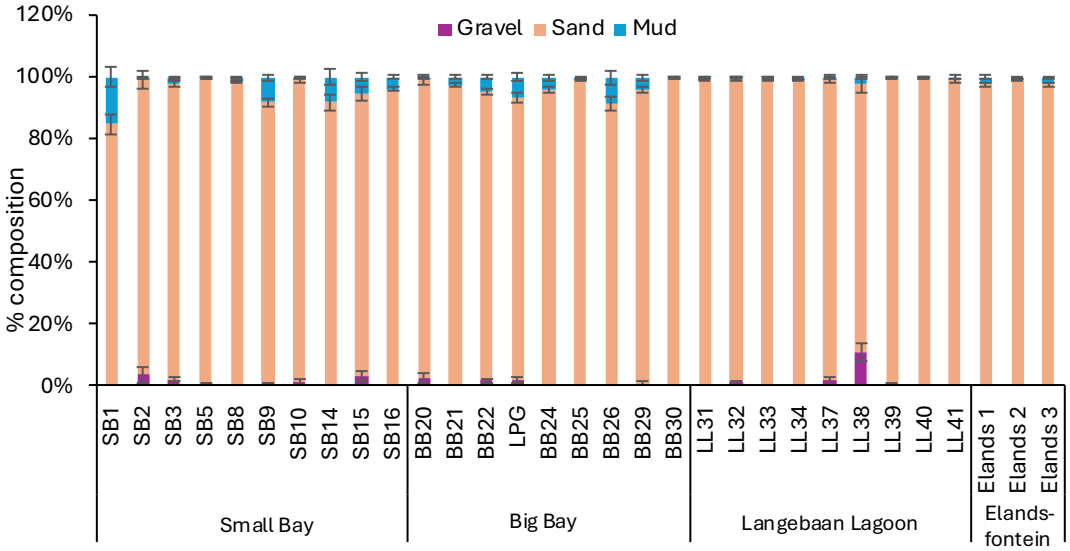


Figure 8.6. Average (±SE) percentage composition of gravel sand and mud in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys.

The spatial distribution of organic matter generally mirrored the patterns observed in the particle size distribution results. Small Bay had the highest average TOC and TON concentrations, at 0.62% and 0.11%, respectively, followed by Big Bay with an average of 0.51% TOC and 0.10% TON (Figure 8.9). Average organic matter concentrations were lower in Elandsfontein (0.20% TOC and 0.03% TON) and lowest in Langebaan Lagoon (0.15% TOC and 0.04% TON) (Figure 8.9). At a site scale, SB1 again stood out, with 2.62% TOC and 0.41% TON. Sites BB20 and LPG also had relatively higher TOC and TON levels (Figure 8.9).

The average C:N ratio results across all sites ranged from 4.84 at SB5 to 9.83 at Elands2, with most sites falling between approximately 5 and 8 (Figure 8.9). These values are broadly consistent with a predominantly marine or algal source of organic matter rather than from terrestrial plants, which generally have higher C:N ratios. The relatively narrow range among sites suggests that differences in TOC and TON are more likely to reflect the amount of organic matter retained or deposited in that area rather than substantial differences in organic matter source.

Trace metal concentrations in Small Bay were on average higher than the other regions of the bay and tended to show the strongest spatial variability. Site SB1 was again distinguished from the other sites, and had the highest average concentrations of aluminium, arsenic, cadmium, chromium, copper, iron, manganese, nickel and zinc, as well as the second highest concentration of lead (Figure 8.7). SB1 was also the only site with average metal concentrations above the ERL limit. At this site, cadmium was 2.3 times higher than the ERL, and copper was 1.5 times the ERL. No other sites had metals exceeding their ERLs. Furthermore, four out of the five metals assessed for enrichment were significantly enriched at SB1 (Figure 8.8). Only SB5 showed a comparable degree of enrichment, with three metals significantly enriched and one very highly enriched; this site also exhibited the greatest enrichment of copper and nickel among all sites assessed (Figure 8.8).

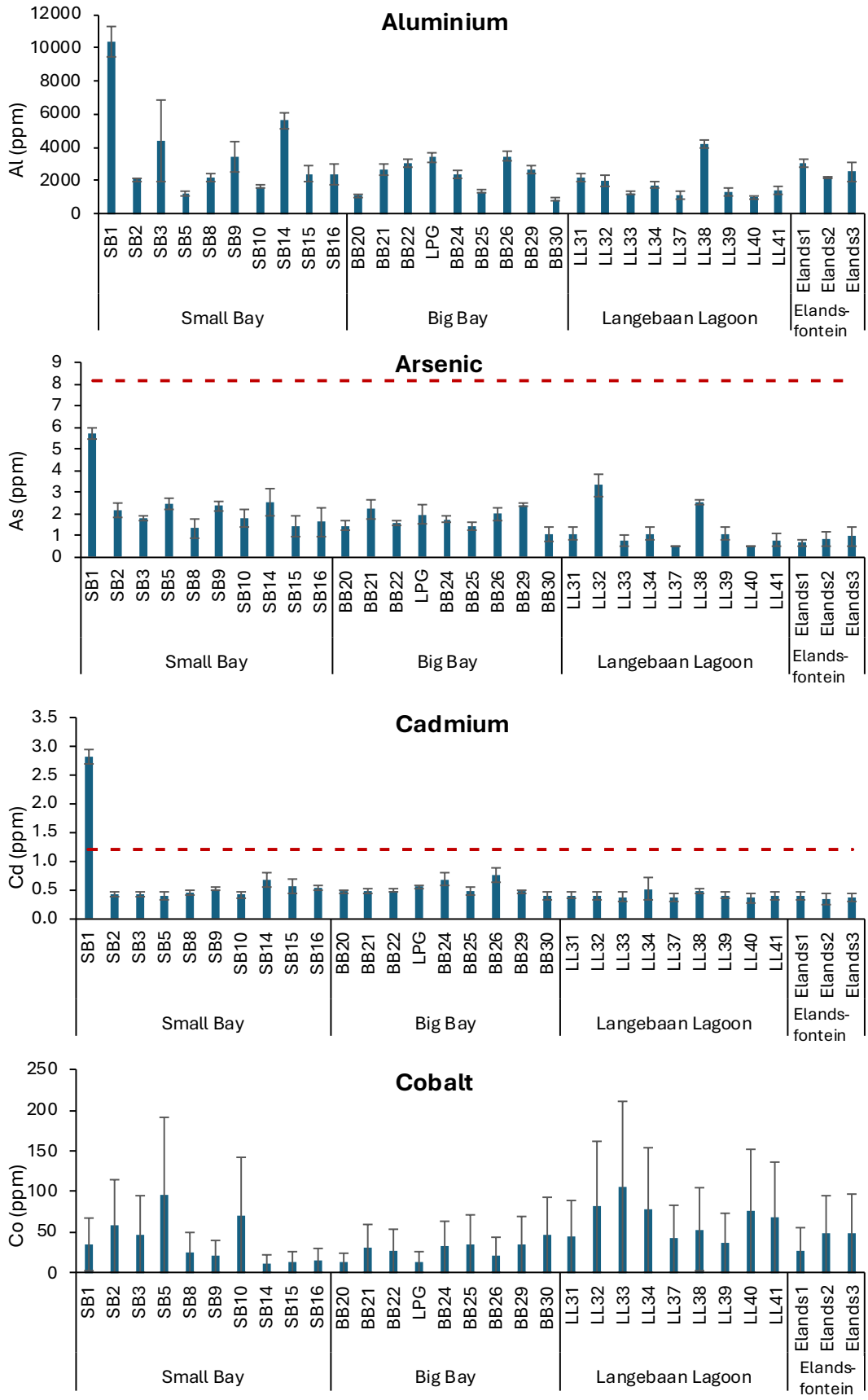


Figure 8.7. Average (±SE) trace metal content in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys. Red dashed lines indicate the NOAA Effects Range Low limit.

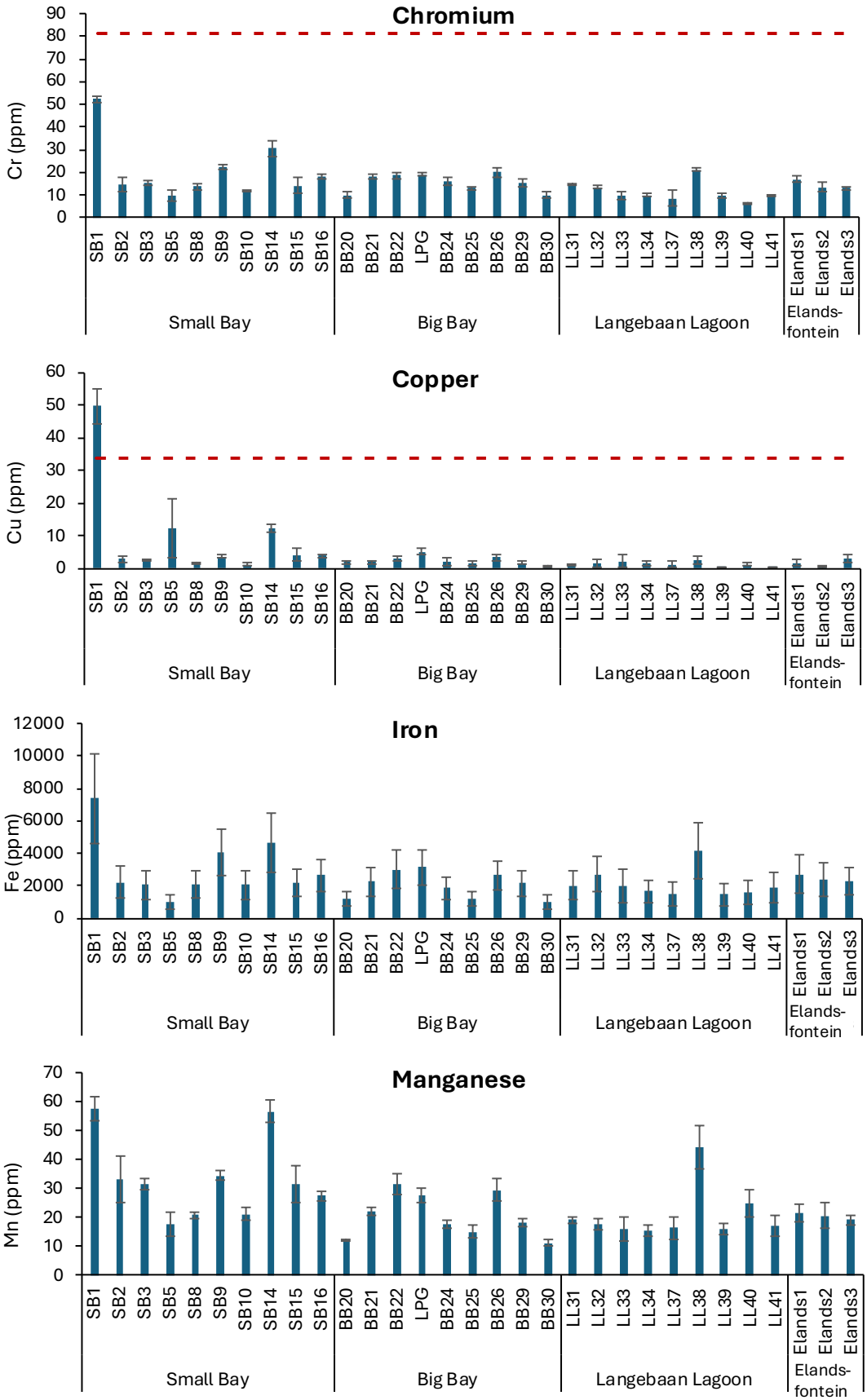


Figure 8.7. continued. Average (±SE) trace metal content in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys. Red dashed lines indicate the NOAA Effects Range Low limit.

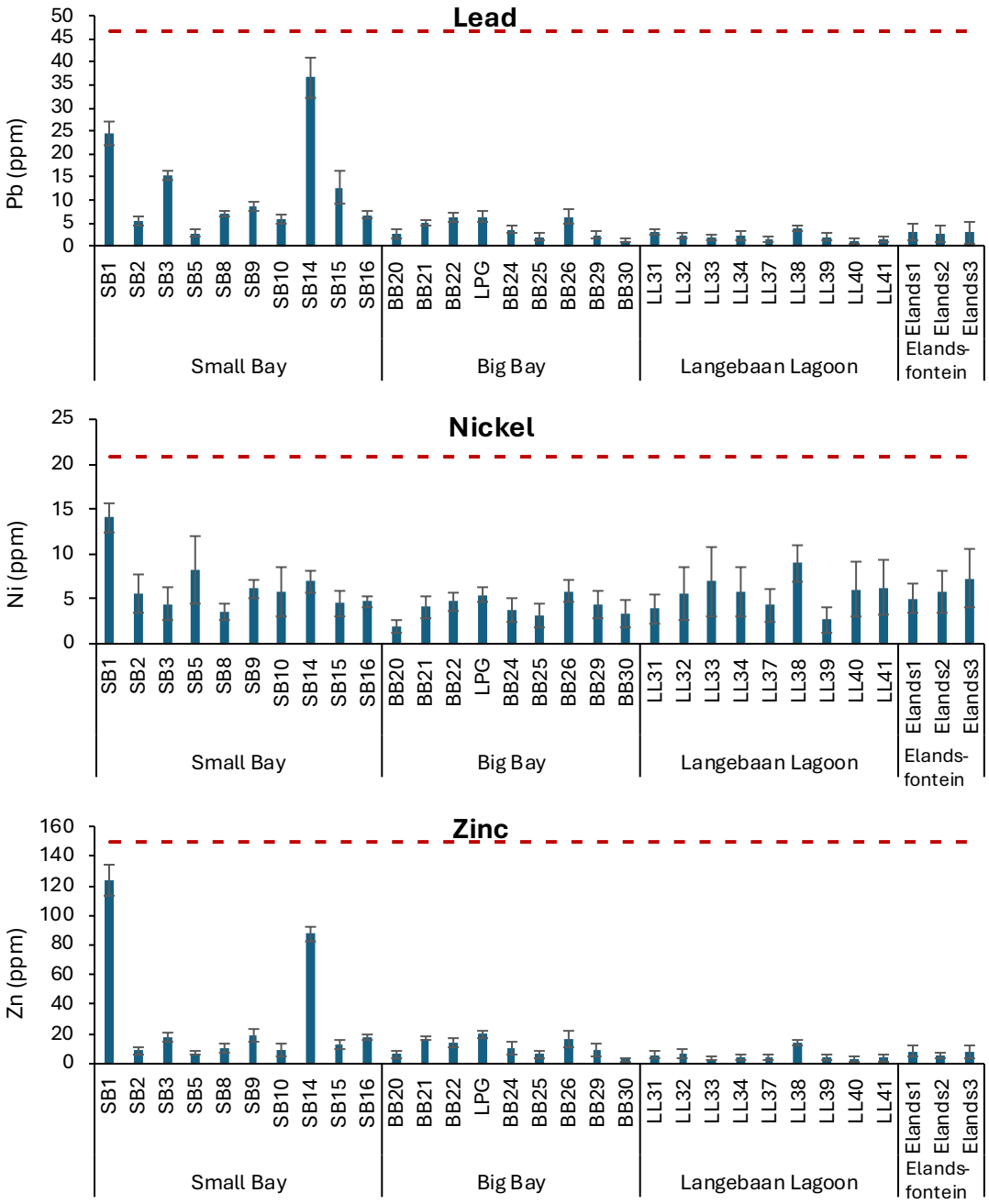


Figure 8.7 continued. Average (±SE) trace metal content in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys. Red dashed lines indicate the NOAA Effects Range Low limit.

Another Small Bay site, SB14, exhibited the second-highest levels of aluminium, chromium, copper, iron, manganese, and zinc, and the highest levels of lead (Figure 8.7). This site also had relatively high levels of mud, at 8.1%, but was not particularly organically enriched (Figure 8.9). Cobalt was not especially high at sites SB1 and SB14 but rather peaked at sites LL33 in the Lagoon and SB5 in Small Bay. SB14 had two significantly enriched metals and one very highly enriched metal, including the highest levels of lead and zinc enrichment, but also had the lowest levels if nickel enrichment (Figure 8.8).

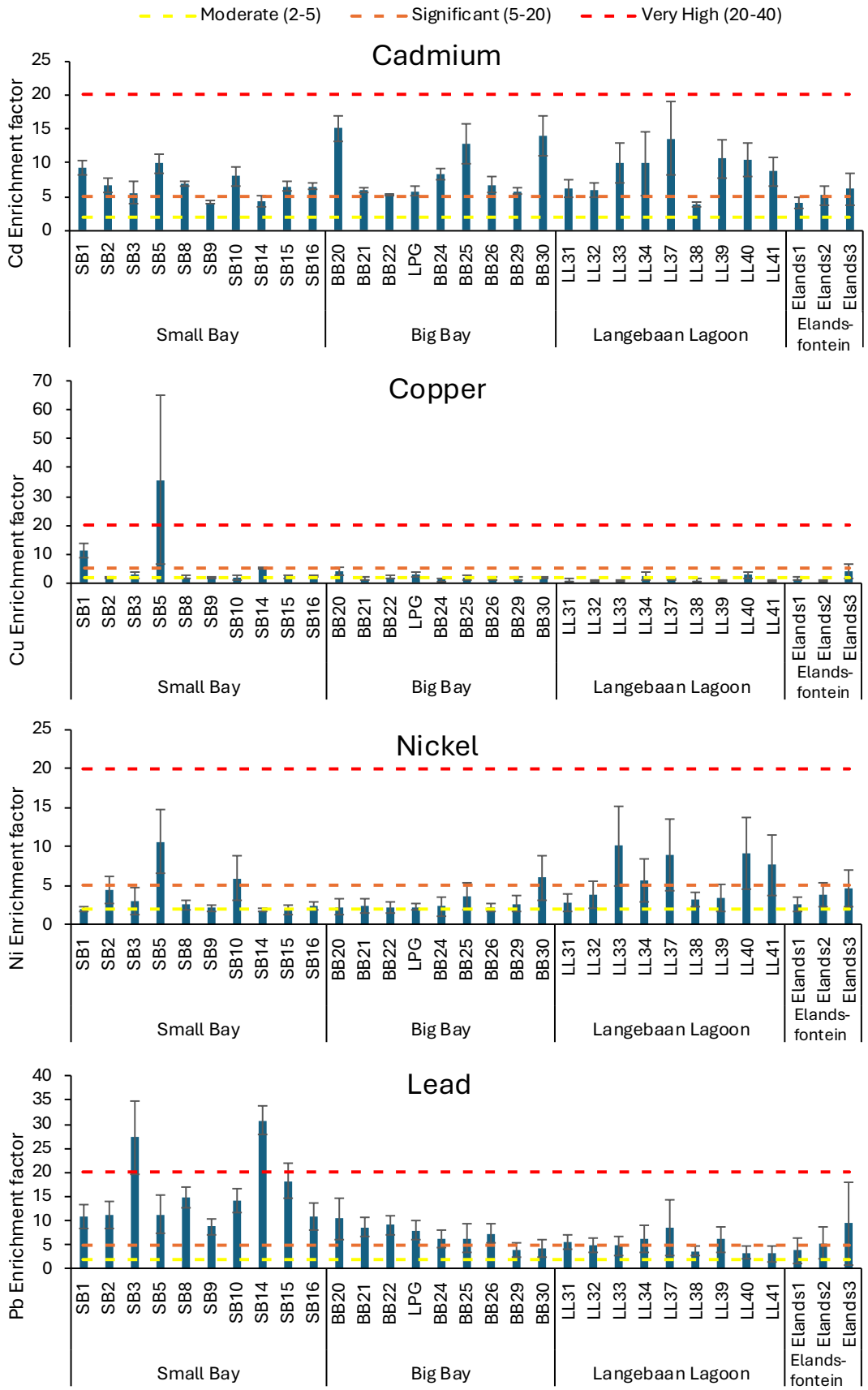


Figure 8.8 Average (±SE) trace metal enrichment in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys. Coloured dashed lines indicate moderate, significant and very high enrichment ranges

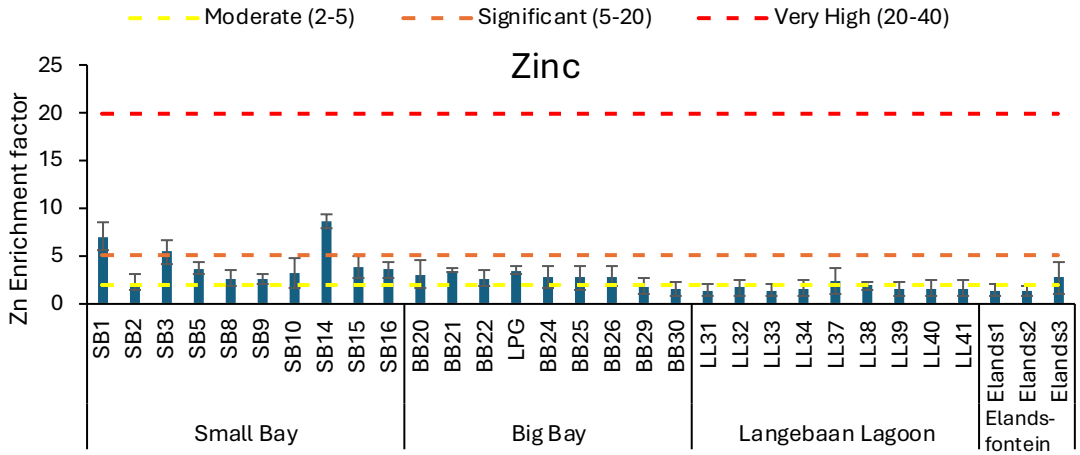


Figure 8.8 continued. Average (±SE) trace metal enrichment in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys. Coloured dashed lines indicate moderate, significant and very high enrichment ranges

The higher trace metal concentrations observed in Small Bay are likely related to its sheltered nature, with low current speeds, limited wave action and reduced water exchange resulting in reduced flushing and greater retention of contaminants within the area. These hydrodynamic conditions are partly associated with anthropogenic modification of the Bay, including the construction of the IOT, breakwaters and Marcus Island causeway, which have altered local wave energy and current patterns (Kruger et al. 2005). The reduced hydrodynamic energy and flushing capacity of Small Bay may promote the deposition and retention of fine sediments and associated organic matter, which can provide additional binding sites for trace metals.

Langebaan Lagoon and Elandsfontein tended to have lower concentrations of trace metals and lower enrichment. Cadmium was least enriched at LL38 and Elands1, copper at LL32 and Elands2, lead at LL40 and LL41, and zinc at LL31 and Elands2 (Figure 8.8).

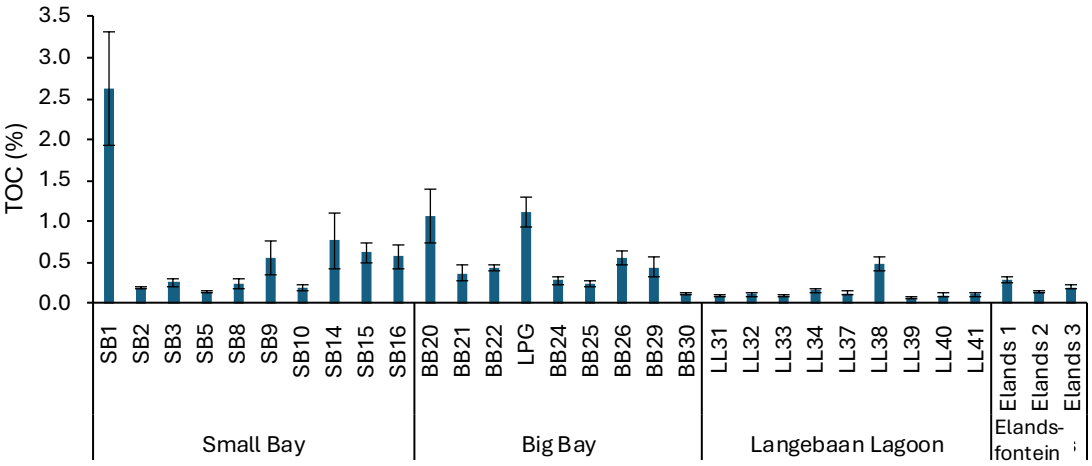


Figure 8.9. Average (±SE) Total Organic Content (top), TON (middle), and Redfield Ratio (C:N) (bottom) in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys.

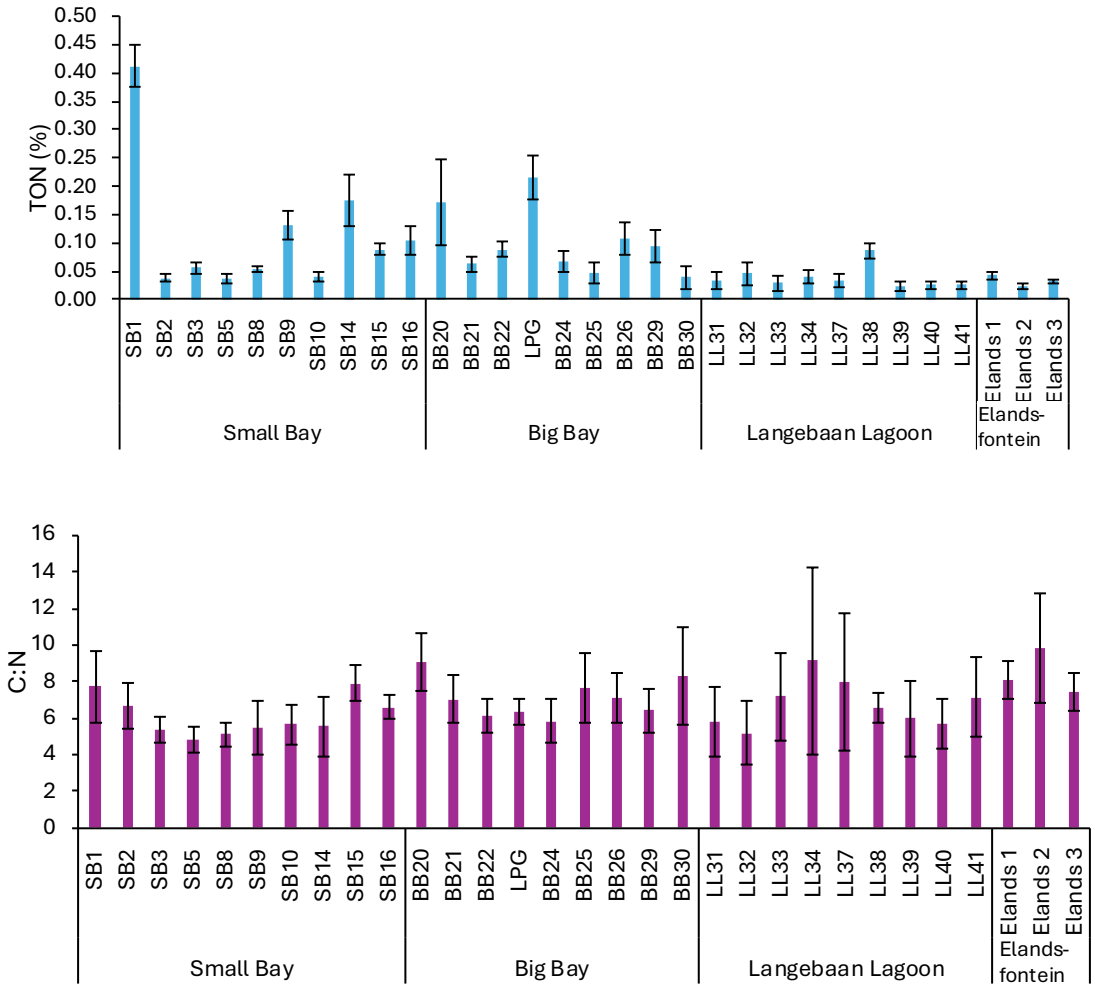


Figure 8.9 continued. Average (±SE) Total Organic Content (top), TON (middle), and Redfield Ratio (C:N) (bottom) in sediment at sites sampled between 2020–2023 and in 2025 during the annual State of the Bay surveys.

8.5 SPATIAL VARIATION IN BENTHIC MACROFAUNA

8.5.1 SPECIES DIVERSITY

Spatial variation in the Shannon Diversity Index (H') from the 2026 survey is presented in Figure 8.10. Diversity was generally highest in Langebaan Lagoon, apart from LL39, and at several sites in Big Bay (BB20, BB24, BB25, BB29, and BB26), where H' values ranged from 2.2 to 2.9. In contrast, the lowest diversity was recorded at two Elandsfontein sites (H' = 1.2), at LPG in Big Bay (H' = 0.9) and at LL39 in Langebaan Lagoon (H' = 1.4). Low diversity was also observed at several sites in Small Bay, particularly around the Iron Ore Terminal (IOT) (SB3, SB14) and within the basin (SB2, SB3, SB5, SB8, SB14), where H' ranged from 1.4 to 1.7. Exceptions included SB1 near the Saldanha Bay Yacht club (H' = 1.9), and SB15 and SB16 near the IOT (H' = 1.9–2.2), which showed intermediate to relatively high diversity.

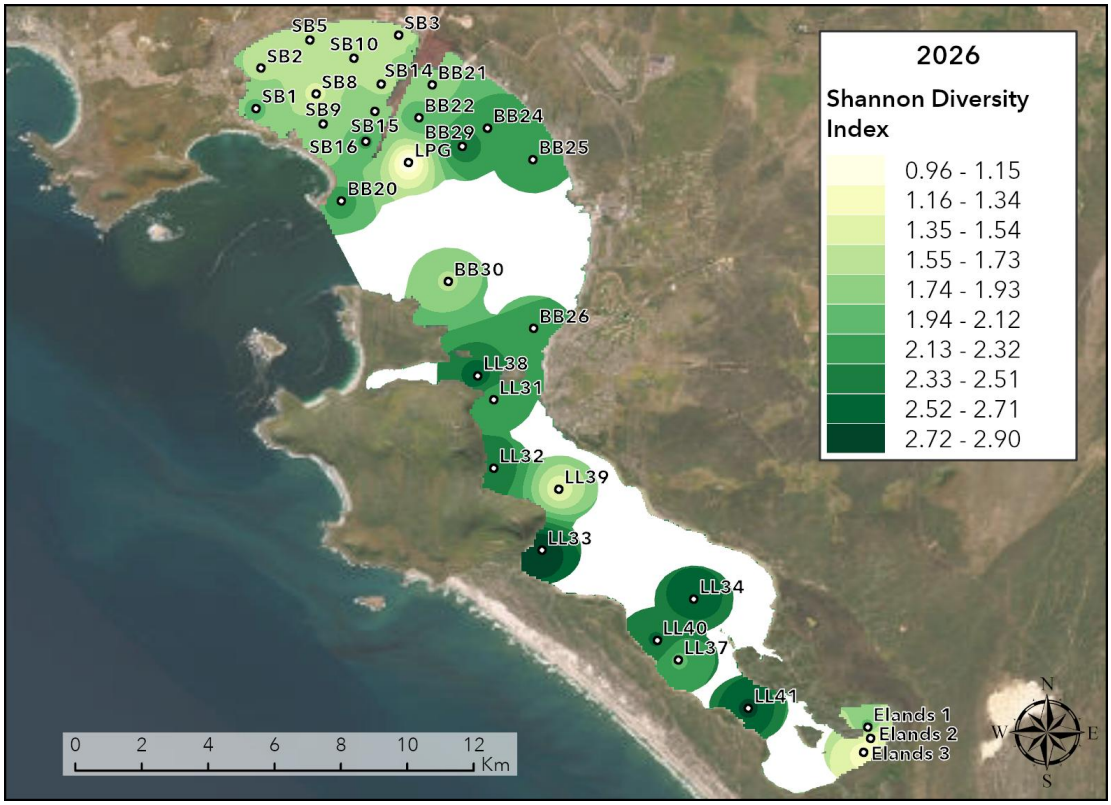


Figure 8.10. Shannon diversity of the benthic infauna at monitoring sites in Saldanha Bay and Langebaan Lagoon as indicated by the 2026 survey results. SB: Small Bay, BB: Big Bay, LPG: Liquid Petroleum Gas berth, LL: Langebaan Lagoon, Elands: Elandsfontein.

The relatively low diversity observed in Small Bay and the IOT may be associated with reduced water exchange, greater retention of fine sediments and organic material, and potentially more frequent hypoxic or anoxic conditions in these sheltered areas, linked to anthropogenic modification of the Bay, including the construction of the IOT, breakwaters and Marcus Island causeway, which have altered wave energy and local current patterns, particularly within Small Bay (Kruger et al. 2005). This has led to reduced wave energy and modified current patterns, especially in Small Bay. Although natural temporal variability may contribute to the observed patterns, the degree of anthropogenic modification within Saldanha Bay provides a plausible explanation for the consistently lower diversity observed in parts of Small Bay. In Big Bay, LPG exhibited the lowest diversity recorded in the 2026 survey (H' = 0.9). The low diversity at this site may be associated with ongoing physical disturbance related to the Single Point Mooring. Conventional single-point mooring systems can result in repeated interaction between the mooring system and the seabed as the vessel moves in response to wind, waves and tidal conditions. This movement can cause the mooring chain and associated ground tackle to repeatedly sweep or scour the surrounding seabed, resulting in mechanical disturbance of sediments and benthic organisms (Unsworth et al. 2017, Glasby & West 2018, Langley et al. 2026). Unlike discrete disturbance events, such as individual anchor drops, repeated chain movement can represent a more persistent source of disturbance and may limit the establishment or recovery of less disturbance-tolerant benthic taxa.

The drivers of low diversity at LL39 are less certain, although natural disturbance may provide a more plausible explanation. LL39 is located within the central channel, where

relatively high tidal current velocities may result in frequent physical disturbance. Similarly, the low diversity observed at Elandsfontein may be associated with the exposed, shallow sandbank environment, where wave action and associated sediment disturbance are likely to be important influences. Natural disturbances, including wave exposure, elevated turbidity and fluctuations in nutrient availability, can favour opportunistic and short-lived species and limit the establishment of more diverse, stable communities. Similar effects may occur under anthropogenic disturbance, which can also favour disturbance-tolerant and opportunistic taxa.

8.5.2 COMMUNITY STRUCTURE

A multivariate ordination prepared from 2026 infaunal abundance data is presented in Figure 8.11. The ordination reveals patterns broadly consistent with those observed in the diversity data. In particular, the infaunal communities at Elandsfontein are distinct from those at Big Bay and Small Bay, while showing greater similarity to communities in Langebaan Lagoon. Overall, Elandsfontein communities were most similar to those in Langebaan Lagoon (69% dissimilarity), and most distinct from communities in Saldanha Bay (Small Bay and Big Bay), with dissimilarities of up to 91%. These results indicate a clear spatial gradient in community structure, ranging from the more marine-dominated habitats of Saldanha Bay, through the semi-sheltered Langebaan Lagoon, to the highly sheltered, shallow, sun-warmed, and potentially freshwater- or estuarine-influenced habitats at Elandsfontein. This was further supported by statistically significant differences between all four areas (PERMANOVA pseudo-$F_{3,33} = 6.61$; $p = 0.001$). Pairwise comparisons showed significant differences between all sampling locations ($p = 0.001$ for all comparisons), with the greatest differences occurring between Elandsfontein and Small Bay (t = 3.30) and Big Bay and Elandsfontein (t = 3.22). Similar findings were found by (Kruger et al. 2005), between communities recorded at Small Bay and Big Bay, and attributed it primarily to anthropogenic activities altering and the physical division of the bay by the harbour wall, which altered water movement, sediment deposition and pollution and organic enrichments.

The 2026 survey also highlights differences in species sensitivity across sites. “Sensitive” species, intolerant of high levels of anthropogenic disturbance, were prevalent at Elandsfontein and Langebaan Lagoon but largely absent from sites near the Big Bay and Small Bay sites, particularly near the IOT. These species included large bodied polychaetes such as *Notomastus latericeus, Marphysa sherlockae* and *Scolaricia dubia*.

It should be noted that differences in macrofaunal community structure are also partly explained by the physical and environmental parameters present at each site (i.e. freshwater ingress, tidal currents, sediment granulometry and depth).

“Hardier” filter feeders, such as *Upogebia capensis*, are abundant in Small Bay samples, while more “sensitive” filter feeders, like the amphipods *Ampelisca palmata, A. brevicornis,* the mollusc *Macomopsis ordinaria* and the tongue worm *Listriolobus capensis* are more abundant in Big Bay than Small Bay. Additionally, the sea pen *Virgularia schultzei*, a sensitive species, was only found in Big Bay.

The relationship between 2026 macrofaunal abundance data and abiotic data from the five most recent surveys (depth, sediment grain size fractions, TOC, TON and trace metals) was investigated using a distance-based linear model (DistLM) (Anderson et al. 2008). The DistLM was significant, indicating that the measured environmental variables

collectively explained a significant proportion of variation in benthic macrofaunal community composition ($F_{13,17} = 1.28$, $p = 0.022$). Of the individual variables, the C:N ratio ($F_{1,17} = 2.20$, $p = 0.011$), depth ($F_{1,17} = 1.77$, $p = 0.031$), TOC ($F_{1,17} = 1.63$, $p = 0.036$), and percentage mud ($F_{1,17} = 1.73$, $p = 0.044$) were significant predictors of community composition, while the remaining variables were not significant ($p > 0.05$). In total, the dbRDA axes captured 69.6% of the variation explained by the environmental variables, with dbRDA1 explaining 50.2% and dbRDA2 explaining 19.4%. These findings are consistent with regional and international studies from comparable coastal environments, which similarly demonstrate that mud fraction, organic content and depth are key abiotic drivers of benthic infaunal community structure and behaviour in bays and lagoons worldwide (Harmer 2014, Robertson et al. 2015, Aliakbarian et al. 2020, Wiesebron et al. 2021). The dbRDA plot indicated a clear separation of macrofaunal communities among sampling regions, primarily along dbRDA1 (Figure 8.12). Small Bay and Big Bay samples were generally associated with the depth and TOC gradients, while Langebaan Lagoon and Elandsfontein samples were more closely associated with the mud content and C:N gradients.

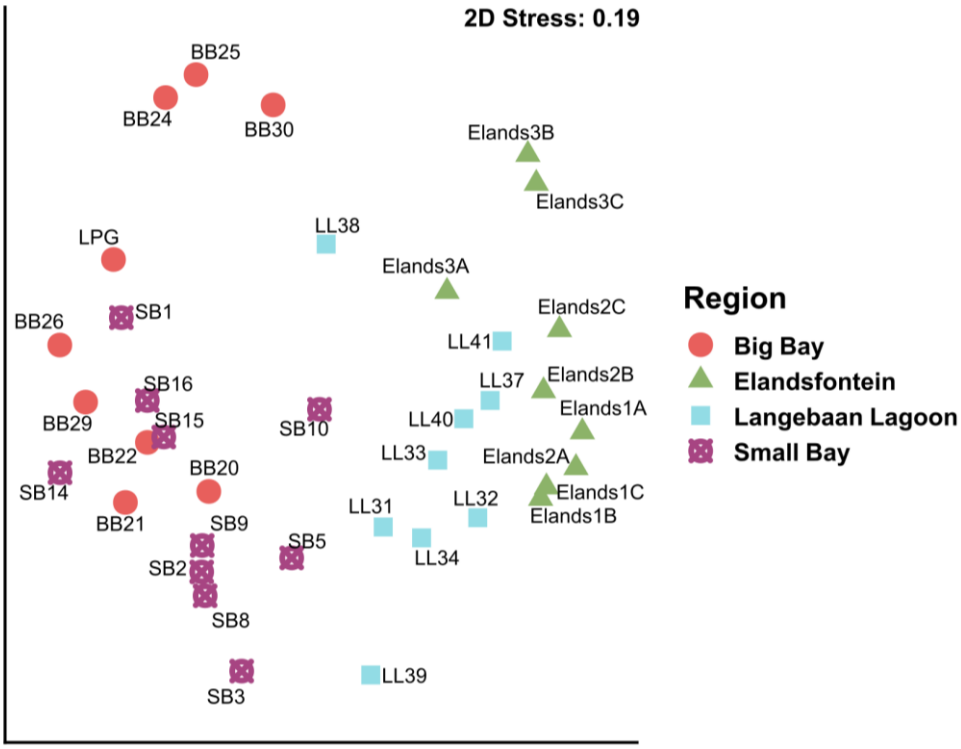


Figure 8.11 Similarity among sample sites based on benthic infauna abundance in 2026. Areas included are: Small Bay (SB), Big Bay (BB), Langebaan Lagoon (LL) and Elandsfontein (Eland).

Within Small Bay, SB1 and SB14 were separated from the other sites in the dbRDA plot, highlighting considerable spatial variability in macrofaunal community composition within the bay. Their separation was associated primarily with the TOC gradient, which was identified as a significant predictor of community composition in the DistLM analysis. The distinct position of these sites may be related to elevated organic loading associated with nearby anthropogenic activities, including effluent and organic waste from fish-processing facilities and mussel culture. Sediment data indicate elevated TOC, fine sediment and trace metal concentrations in this area (Figure 8.6, Figure 8.7, Figure

8.8 and Figure 8.9), particularly at SB1. Reduced water exchange within Small Bay may further promote the deposition and retention of fine sediments, organic material and associated contaminants, potentially contributing to the distinct macrofaunal communities observed at these sites (Kruger et al. 2005).

Key species contributing to the dissimilarity between the Saldanha Bay and Langebaan Lagoon samples include the coastal mudprawn *Upogebia capensis,* the filter feeding amphipods *Ampelisca* sp.*,* and the predatory whelks *Nassarius* sp., all of which were relatively abundant in Small Bay and Big Bay but rare or absent in lagoon samples. Conversely, species such as the sand prawn *Kraussillichirus kraussi,* the isopod *Natatolana hirtipes*, the crown crab *Hymenosoma orbiculare* and polychaete *Orbinia angrapequensis* (detritivores, scavengers or predators) were more abundant in the Lagoon.

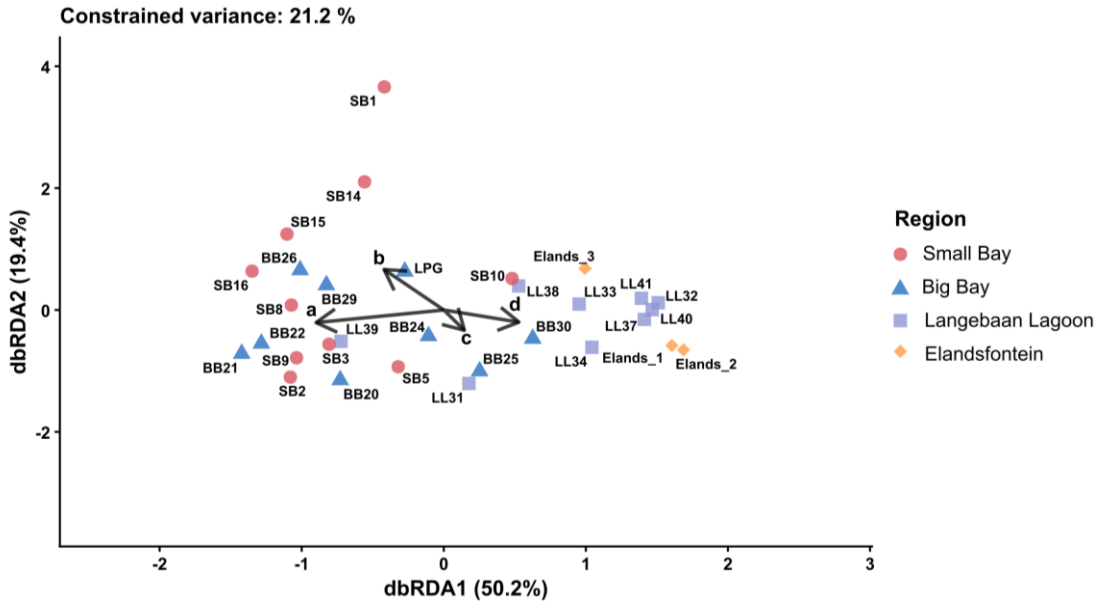


Figure 8.12. dbRDA plot of 2026 macrofaunal abundance data. Significant predictors are displayed, namely (a) depth (m), (b) TOC, (c) % mud and (d) C:N ratio. The black lines represent these predictors as vectors, with vector length indicating the strength of the relationship between each predictor and the axes.

The benthic infaunal community at Elandsfontein was dominated by small crustaceans, mostly amphipods, and polychaetes (Figure 8.13). The presence of unique species such as the sandflat crab *Danielella edwardsii* and the high abundance of the sand prawn *Kraussillichirus kraussi*, the mud prawn *Upogebia africana*, and polychaete *Marphysa sherlockae* were the primary factors driving the dissimilarity between Elandsfontein and the Saldanha Bay and Langebaan Lagoon communities.

Species composition is often more easily understood when examined at higher taxonomic or functional group levels (e.g., feeding mode). Infaunal abundance and biomass for each area sampled are shown in Figure 8.13. Crustaceans (including prawns, shrimps, mysids, crabs, amphipods, and isopods) were the dominant taxonomic group in all areas, followed by polychaetes (bristle worms). Filter feeders dominated the functional groups in all areas, followed by detritivores (Figure 8.13). These differences

are largely attributed to physical habitat variations across the sampled areas, which are influenced by past and ongoing anthropogenic activities such as port construction, dredging and pollution (Kruger et al. 2005). In general, overall biomass was lower at Elandsfontein compared to the Bay and Lagoon sites.

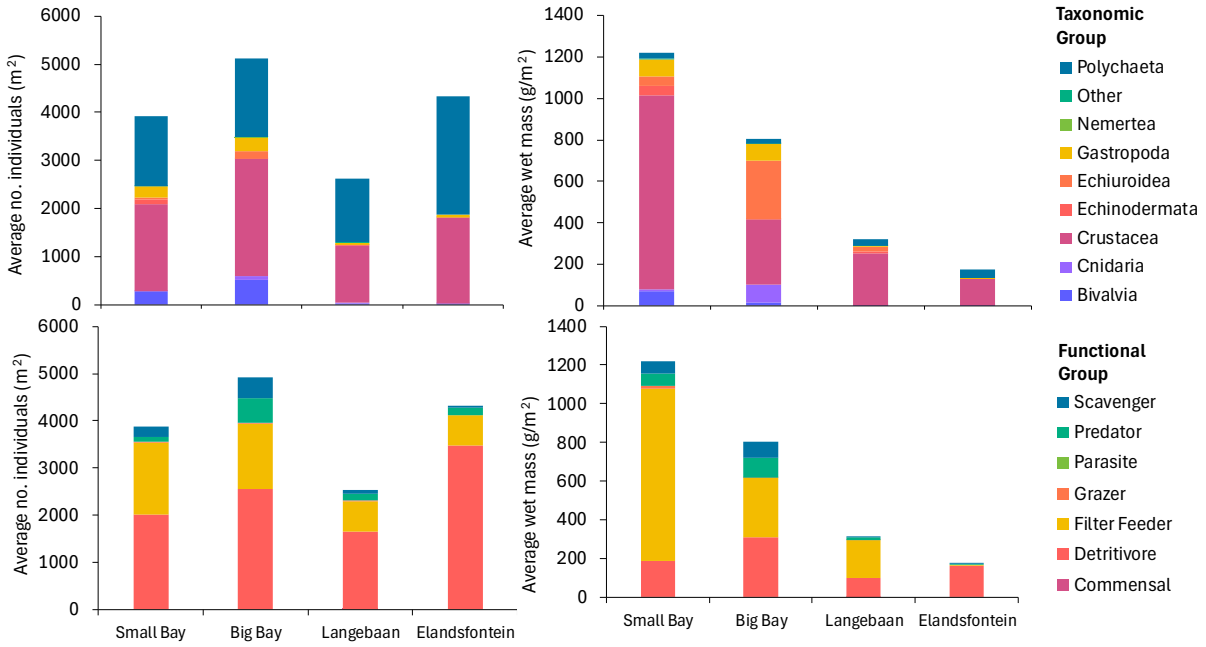


Figure 8.13. Average abundance and biomass (g/m^2) of benthic infauna by functional and taxonomic group in Big Bay (n=9), Small Bay (n=10), Langebaan Lagoon (n=9), Elandsfontein (n=15) in 2026.

8.6 TEMPORAL TRENDS IN BENTHIC MACROFAUNA

8.6.1 SPECIES DIVERSITY

Variation in the total number of infaunal species recorded in Small Bay, Big Bay, Langebaan Lagoon and Elandsfontein during each annual survey from 1999 to 2026 is presented in Figure 8.14. Small Bay has experienced a slight decrease in species numbers over time, however Species numbers fluctuate considerably between years, but there is no clear long-term linear trend as the regression explains only 0.23% of the variation. In contrast, Big Bay strongest positive trend is observed here, with an estimated increase of 1.1351 species per year. The linear trend explains approximately 41.6% of the variation in species numbers. Although there are annual fluctuations, the overall pattern suggests increasing species richness. Langebaan Lagoon and Elandsfontein both show slight increase in species numbers per year, with substantial fluctuations in species numbers.

When interpreting these data alongside recent developments in the Bay, it becomes clear that changes in species richness may be linked to major dredging events. These include the removal of 2 million m^3 of material in 1996/97 for the construction of the Multi-Purpose Terminal, 50 000 m^3 in 2007/08 near the Mossgas quay, and 7 300 m^3 in 2009/10 from the Small Bay side of the IOT. Species richness typically declines immediately

following these dredging events (e.g., 1999, 2008 and 2012) but tends to recover or increase during the intervening periods (Figure 8.14).

The low species richness in Langebaan Lagoon during the 2004 survey may be due to a different phenomenon. During the mid-1990s, the alien invasive mussel *Mytilus galloprovincialis* began establishing dense intertidal beds on two sand flats near the mouth of the Lagoon (Hanekom & Nel 2002). By 1999, these mussel beds reached an estimated biomass of nearly 8 tonnes, raising concerns about the invasion spreading to other parts of the Lagoon (Hanekom & Nel 2002). However, by early 2001, the mussels started to die off, and by mid-2001, only dead shells and anoxic sediment remained. SANParks initiated the removal of dead mussel shells in late 2001 to prevent the re-settlement (Robinson et al. 2007). While the exact cause of the die-off is unclear, siltation and reduced food availability are suggested as potential factors (Hanekom & Nel 2002). It is highly likely that the reduced infauna species richness in the 2004 SOB samples was a residual impact of the mussel invasion.

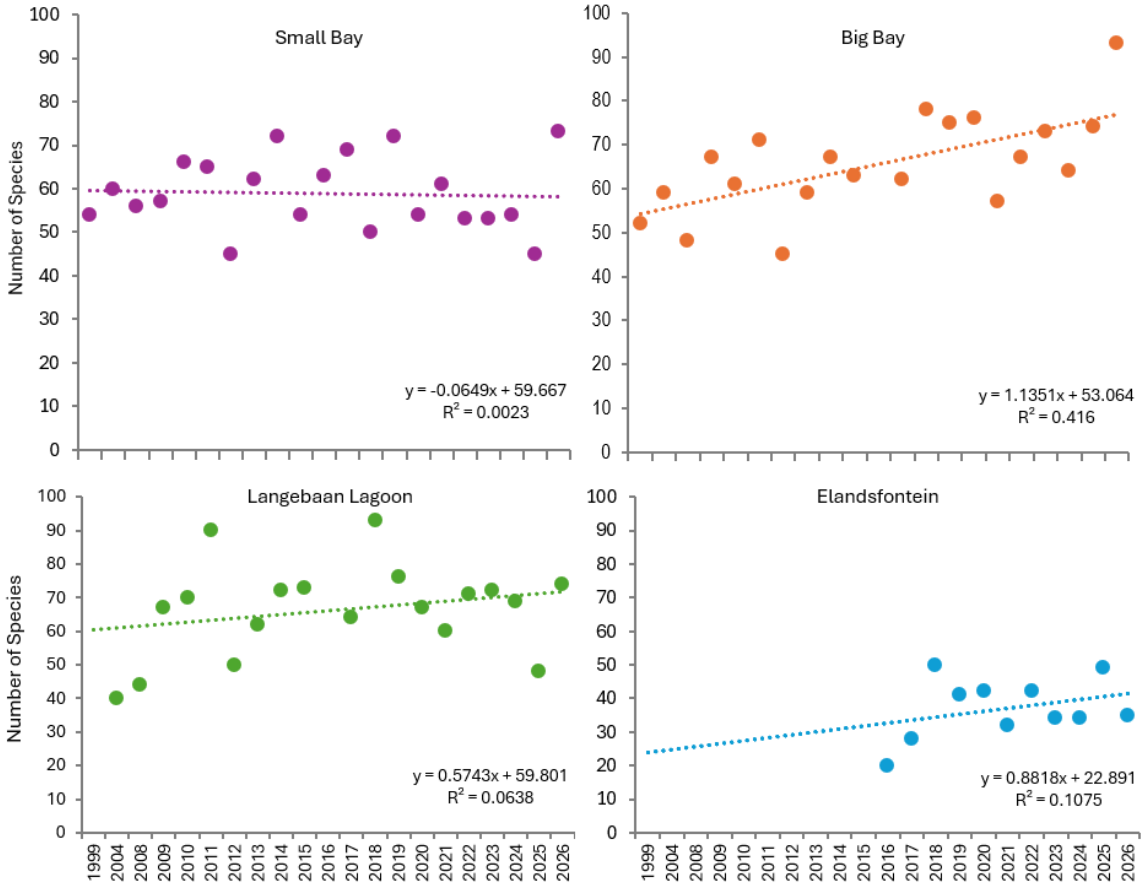


Figure 8.14. Number of species recorded at Small Bay, Big Bay, Langebaan Lagoon (1999–2026) and Elandsfontein (2016–2026).

In the most recent surveys, a decrease in observed species richness was recorded in 2025, particularly in Langebaan Lagoon and Small Bay. However, this apparent decline is likely related to the reduced sampling effort in 2025 rather than an actual ecological shift. When standardised per sample, species richness in 2025 was comparable to, and in some cases higher than, previous surveys. In 2026, species richness increased at several sites across the Bay Area and Langebaan Lagoon, with Big Bay reaching its highest

recorded value of approximately 93 species. In contrast, Elandsfontein has consistently supported lower species richness than the other areas within the system. This may be related to the highly dynamic intertidal mudflat environment, where natural fluctuations in conditions such as temperature and salinity can result in frequent disturbance and favour a less diverse, more opportunistic community. Notably, substantially higher species richness was recorded at Elandsfontein between 2018 and 2022. This pattern was initially considered to be associated with the change in sampling equipment from a hand corer to a Van Veen grab. However, the reintroduction of the hand corer in 2024 has produced species richness values comparable to those obtained using the Van Veen grab, suggesting that both methods can effectively capture species diversity at this site. Nevertheless, differences in sampling equipment and effort may have contributed to some of the observed temporal variation in abundance, biomass and species richness.

8.6.2 Abundance, biomass and community composition

Photographs of benthic infaunal species frequently occurring in Saldanha Bay and Langebaan Lagoon are presented in Figure 8.15. Changes in the abundance and biomass of benthic infauna across Small Bay, Big Bay and Langebaan Lagoon from 1999 to 2026 are presented in Figure 8.16, Figure 8.17 and Figure 8.18, respectively. These figures also highlight the shifting roles of various feeding groups and taxonomic groups over time, offering insights into food availability and species diversity.

Dredging events in 2008, 2012, 2015 and 2017 typically resulted in decreased abundance and biomass, though the infaunal communities exhibited rapid recovery in years following these disturbances. The relative contribution of major taxonomic groups (e.g., Polychaeta, Crustacea, Gastropoda, etc.) shifted during periods of reduced abundance and biomass, but the changes in the feeding groups were more pronounced.

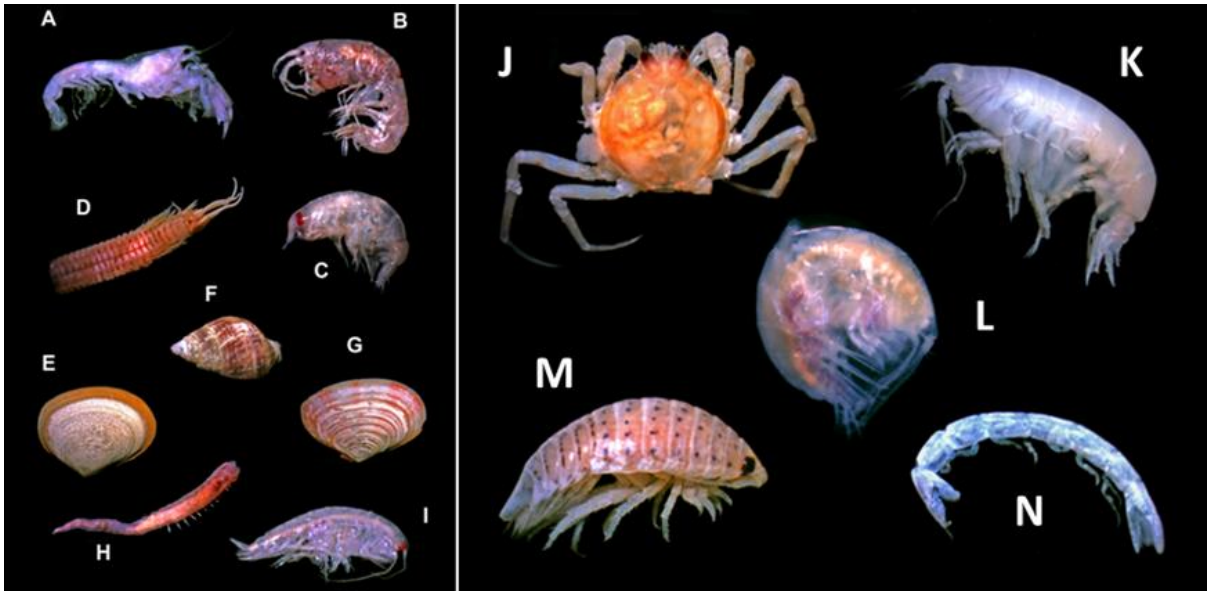


Figure 8.15. Benthic infauna species frequently found to occur in Saldanha Bay and Langebaan Lagoon. (A) *Upogebia capensis*; (B) *Idunella* lindae; (C) *Hippomedon normalis*; (D) *Diopatra monroi*; (E) *Macoma c. ordinaria*; (F) *Nassarius vinctus*; (G) *Tellina gilchristi*; (H) *Sabellides luderitzi*; (I) *Ampelisca anomola*; (J) *Hymenosoma obiculare*; (K) *Socarnes septimus*; (L) *Ampelisca palmata*; (M) *Eurydice longicornis* and (N) *Centrathura caeca*. Photographs by: A. Biccard.

In Langebaan Lagoon, a slight increase in both biomass and abundance was recorded in 2020, driven by the greater presence of sand prawns *Kraussillichirus kraussi,* a species heavily harvested for bait (Figure 8.18). Encouragingly, both abundance and biomass of this species increased, signalling a positive trend for Langebaan Lagoon. However, filter feeders (species that filter particulate matter from the water column) showed a dramatic decline during periods of disturbance, particularly in response to dredging across all three areas of the Bay). In contrast, detritivores, which feed on organic matter, tended to increase during these times.

The greater sensitivity of filter feeders to elevated suspended sediment concentrations may help explain their decline during major dredging activities. The filter-feeding community is largely represented by the mud prawn *Upogebia capensis*, small amphipod species of the genus *Ampelisca* and the sea pen *Virgularia schultzei*. *Virgularia schultzei*, which was previously abundant in both Big Bay and Small Bay, is now absent from Small Bay but persists in Big Bay at low abundances. The loss of this species from Small Bay may therefore reflect reduced habitat suitability associated with the increased mud content and organic enrichment of the sediment, together with altered water-flow patterns within the Bay.

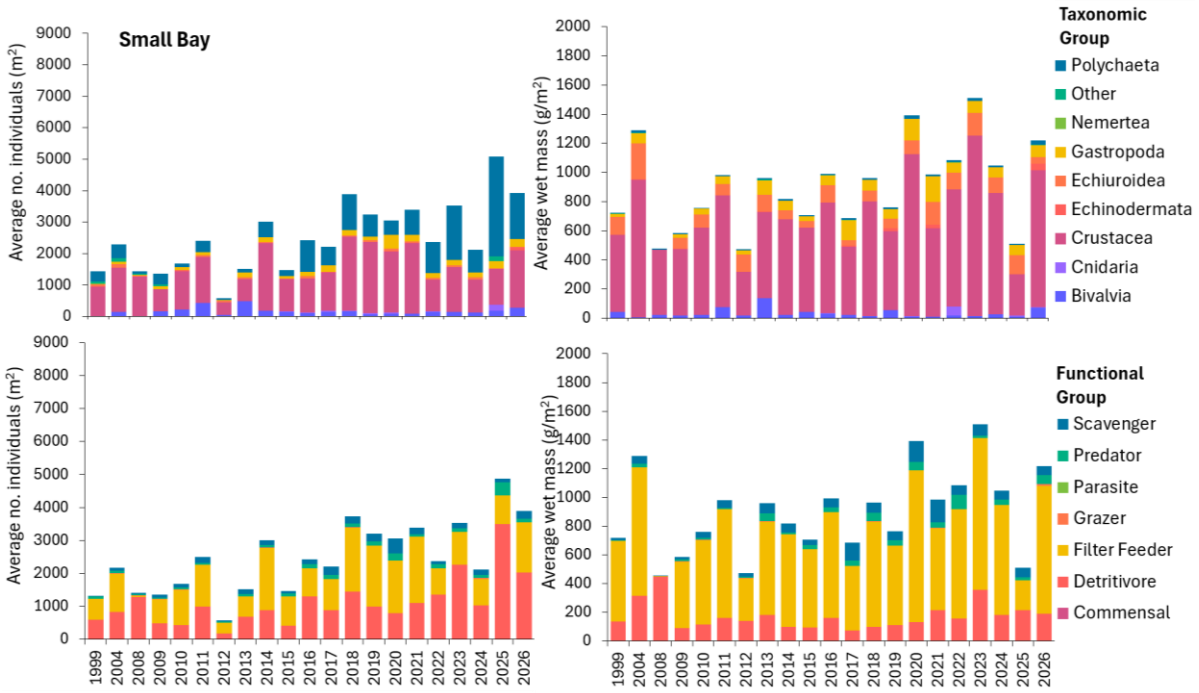


Figure 8.16. Overall trends in the abundance (ave. no. ind./m^2) and biomass (g/m^2) of benthic infauna in Small Bay as shown by taxonomic and functional groups.

Detritivores, the second most dominant group of benthic infauna in Small Bay, are mainly comprised of tongue worms *Listriolobus capensis* and polychaetes from the genera *Polydora* and *Mediomastus*. These taxa are generally more tolerant of variable environmental conditions and may become more abundant or dominant under disturbed or organically enriched conditions. Their increased contribution during periods of disturbance therefore contrasts with the decline observed among the more sensitive filter-feeding taxa.

Following a peak in 2023, both abundance and biomass declined in 2024, with abundance subsequently increasing in 2025 and 2026 (Figure 8.16, Figure 8.17 and Figure 8.18). In 2025, system-wide results indicate increased abundance but reduced biomass across the whole Bay Area and Langebaan Lagoon (Figure 8.16, Figure 8.17 and Figure 8.18). The combination of higher abundance and lower biomass may indicate a greater contribution of smaller-bodied, opportunistic taxa, consistent with a community responding to environmental disturbance. However, this interpretation should be treated cautiously, as changes in sampling effort and environmental conditions may also have contributed to the observed pattern.

The 2026 results showed some increases in macrofaunal biomass at Small Bay and Big Bay relative to several recent surveys. Where biomass increased simultaneously with abundance, this may indicate that the increase in individuals was accompanied by a greater contribution from larger-bodied taxa (Figure 2.17). However, increases were not consistent across all areas or indices, with average abundance at Small Bay, for example, decreasing in 2026. Importantly, the observed abundance and biomass values remained within the range of variation recorded across previous surveys. Thus, although some increases were observed in 2026, the indices can be considered broadly stable within the context of their established temporal variability, with no values exceeding the range previously recorded.

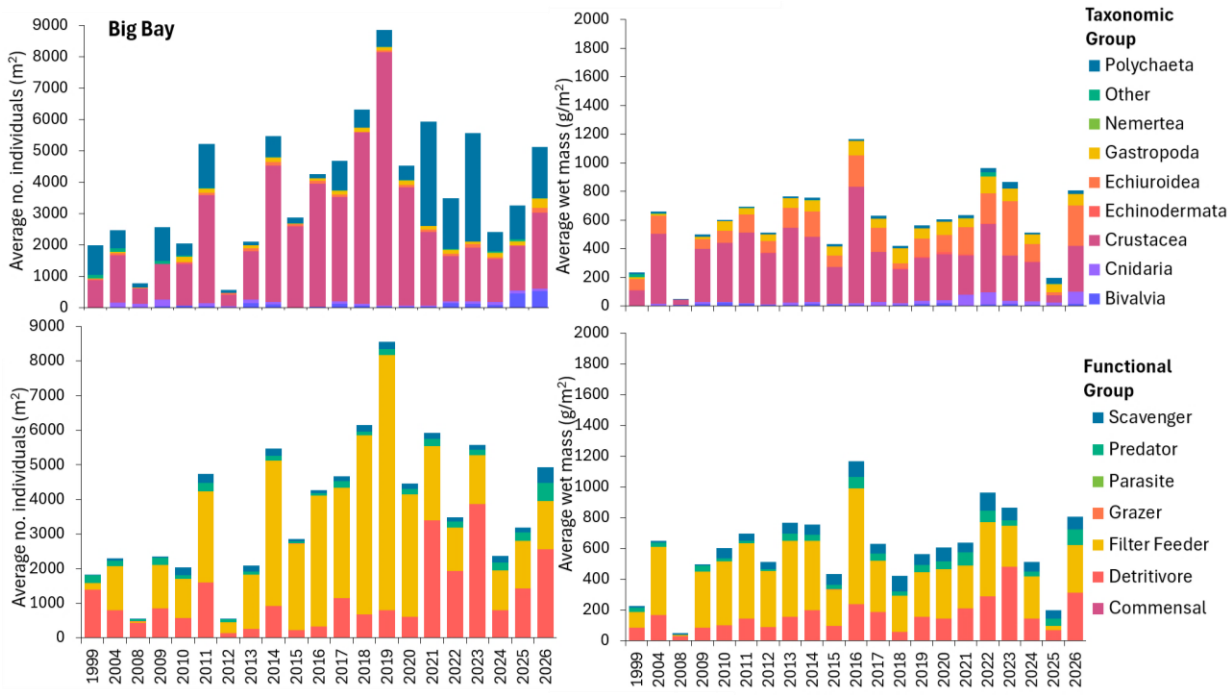


Figure 8.17. Overall trends in the abundance (ave. no. ind./m^2) and biomass (g/m^2) of benthic infauna in Big Bay as shown by taxonomic and functional groups.

Figure 8.18. Overall trends in the abundance (ave. no. ind./m^2) and biomass (g/m^2) of benthic infauna in Langebaan Lagoon as shown by taxonomic and functional groups.

Langebaan Lagoon showed a slight decline in both macrofaunal abundance and biomass in 2026 relative to recent peaks, with biomass decreasing markedly from the high values recorded in 2022–2024. It is important to note that sediment characteristics, which are key determinants of benthic community structure, were not measured during the 2026 survey. Consequently, our ability to interpret the observed patterns in macrofaunal abundance, biomass, and community composition in relation to underlying sedimentary conditions is limited.

Community-wide trends in benthic macrofaunal abundance and biomass differed across the bay and lagoon areas (Figure 8.19). Abundance increased over time at all three locations, with more modest increases in biomass at Small Bay and Big Bay. Langebaan Lagoon showed more pronounced increases in both abundance and biomass over the monitoring period. Interannual variability was evident in both metrics across all locations.

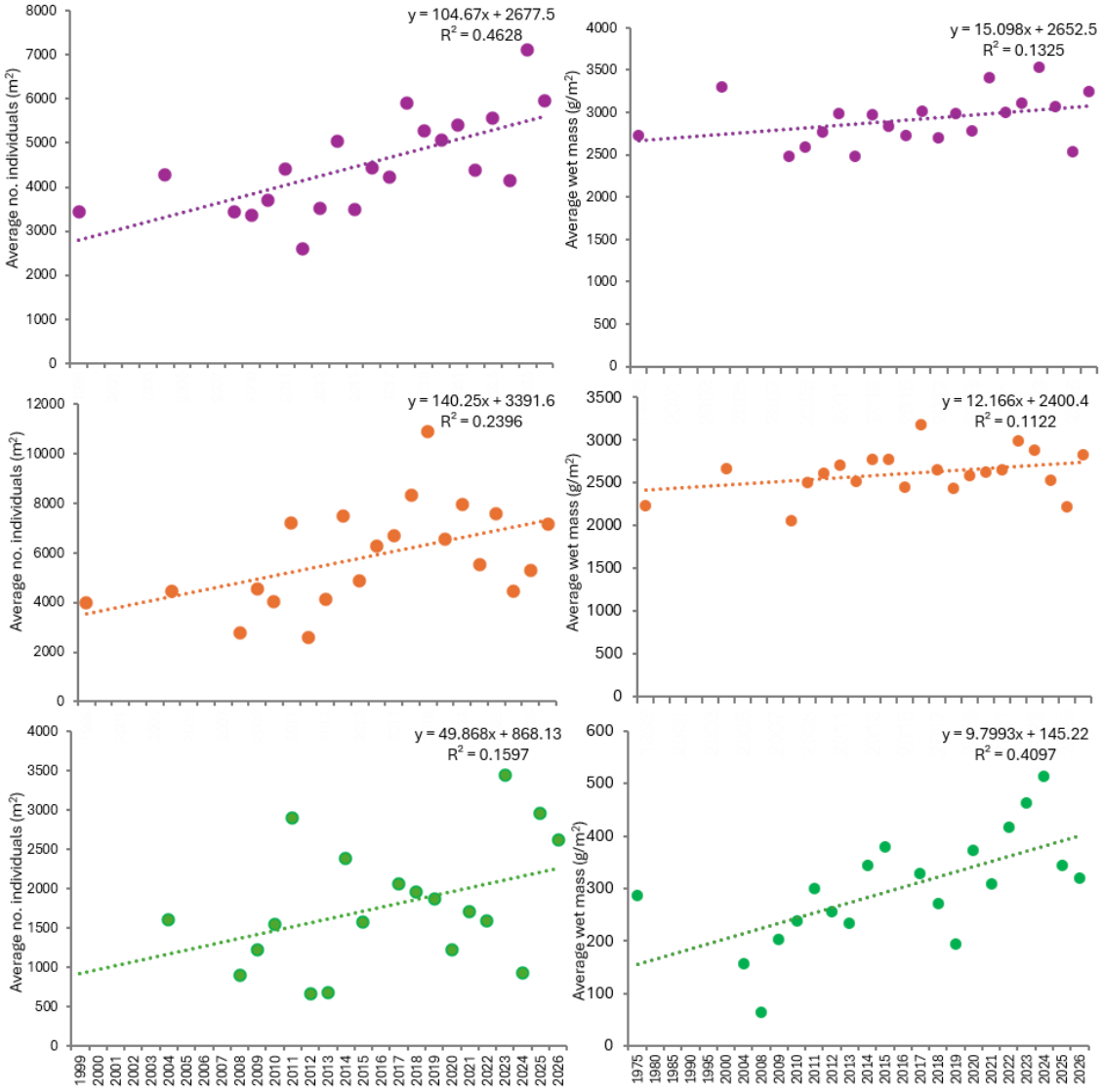


Figure 8.19. Temporal trends in benthic infaunal abundance (ave. no. ind./m2) and biomass (g/m2) across the entire benthic community at Small Bay (top), Big Bay (middle) and Langebaan Lagoon (bottom).

8.6.3 COMMUNITY STRUCTURE

Historical and current reports consistently show distinct differences in infaunal communities across Small Bay, Big Bay, Langebaan Lagoon, and Elandsfontein, mainly due to variations in physical habitat characteristics. To monitor changes in ecosystem health and detect anthropogenic impacts, separate multivariate analyses of infaunal abundance data have been conducted for each area since 2004, avoiding the confounding effects of observed spatial differences.

SMALL BAY

The ordination for Small Bay shows a distinct community shift in 2008, due to high abundances of detritivores, such as the shrimp *Betaeus jucundus*, polychaetes *Mediomastus capensis* and *Maldanidae* sp., and crustaceans from the taxon Cumacea, which were uncommon in other years (Figure 8.20). Conversely, SIMPER results revealed that detritivorous crustaceans such *Spiroplax spiralis*, polychaetes *Polydora* sp. and *Orbinia angrapequensis*, the tongue worm *Listriolobus capensis*, scavenger whelks of the

genus *Nassarius*, filter feeding amphipods *Ampelisca* sp. and the mud prawn *Upogebia capensis*, were common in other years, but were rare or absent in 2008. These observations were likely due to extensive dredging in 2007 and early 2008, which temporarily shifted the community towards more tolerant species. From 2009 to 2026, the community remained stable, with no significant impacts from later dredging events, showing natural variation in species abundance and biomass.

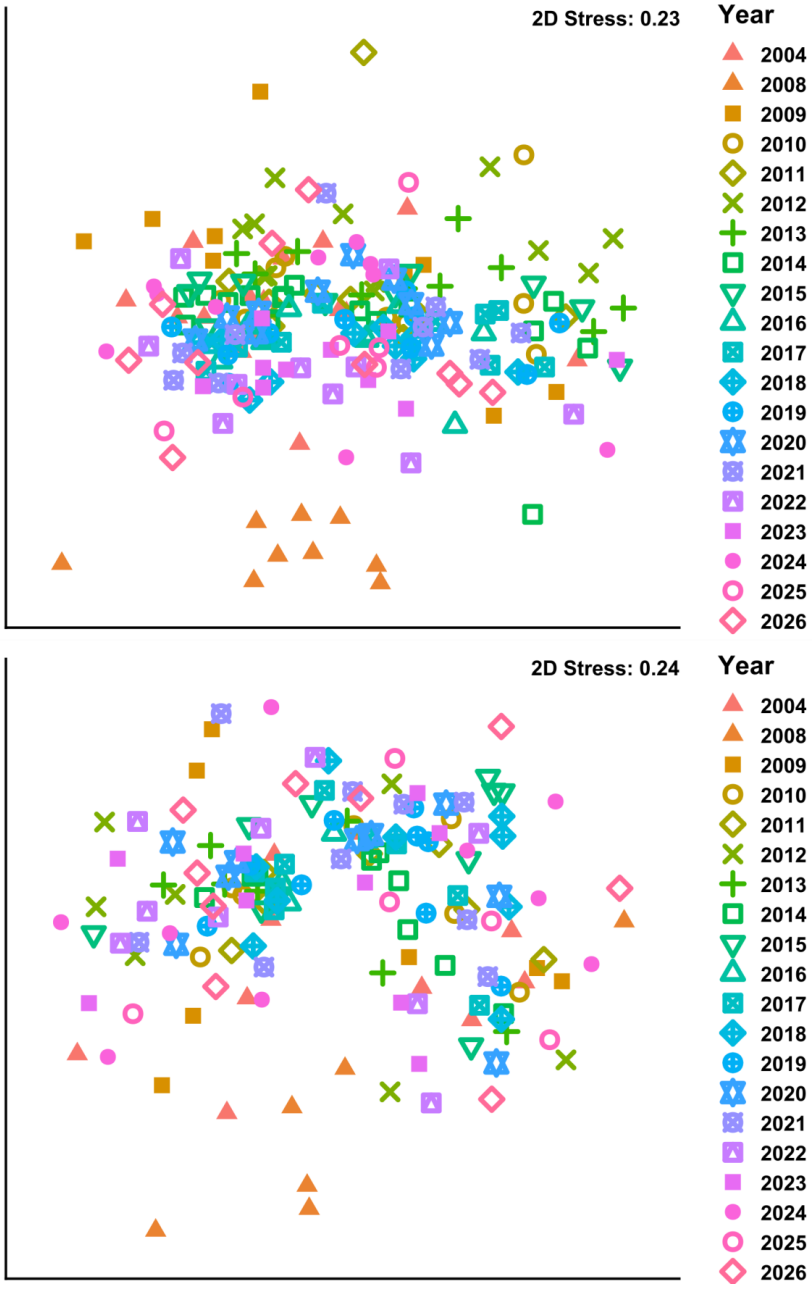


Figure 8.20. nMDS plots based on infaunal abundance data from samples collected in Small Bay (top), Big Bay (bottom) during the period 2004–2026.

BIG BAY

In Big Bay, the 2008 macrobenthos samples also clustered separately from other years, indicting its effect on the community (Figure 8.20). Key species absent or in very low

abundance included detritivores, *Orbinia angrapequensis* and *Listriolobus capensis*, as well as filter feeders like *Upogebia capensis*, *Ampelisca* sp., and *Virgularia schultzei* and predators such as *Nassarius* sp. whelks. Resilient species such as *Betaeus jucundus, Mediomastus capensis*, and *Platynereis australis*, were dominant in 2008, aligning with the findings in Small Bay. The 2026 community composition is consistent with the natural shifts observed in previous years.

LANGEBAAN LAGOON

Similar to Big Bay and Small Bay, 2008 samples from Langebaan Lagoon stand out as outliers due to the absence or low abundance of key species like *Upogebia capensis, Notomastus latericeus* and *Natatolana hirtipes*, combined with the high abundance of *Betaeus jucundus, Marphysa sanguine* and *Eteone foliosa* (Figure 8.20). The community composition in 2026 remains within the natural variations observed in previous years.

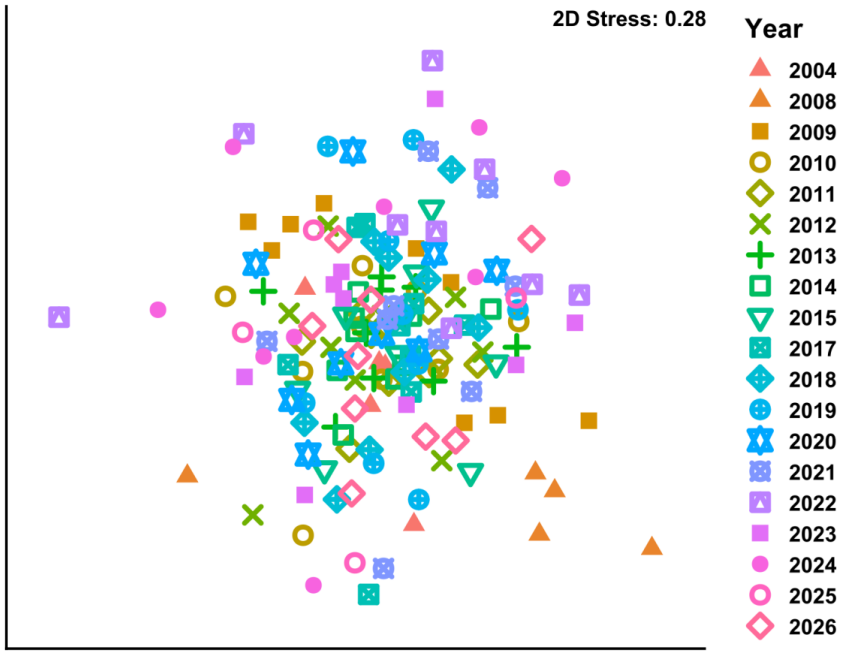


Figure 8.20 continued. nMDS plots based on infaunal abundance data from samples collected in Langebaan Lagoon during the period 2004–2026.

8.7 BENTHIC MACROFAUNA IN ELANDSFONTEIN

Since monitoring began, 81 species of benthic infauna have been recorded at Elandsfontein, with six species unique to this area, including the polychaete *Ancistrosyllis rigida*; the crabs *Danielella edwardsii* and *Paratylodiplax algoensis*; the gastropod *Nassarius kraussianus*; and an isopod belonging to the family Sphaeromatidae.

Infaunal abundance and biomass results from 2016 to 2026, divided into taxonomic and functional feeding groups, is shown in Figure 8.21. While other areas showed fluctuating trends over time, Elandsfontein exhibited a decline in abundance and biomass between 2020 and 2023, the causes for which remains unclear. In 2024 and 2025, however, an abrupt spike was recorded. This increase is most likely linked to a change in sampling

equipment, as the 2024–2025 results align more closely with those from 2016–2017, when a hand corer was used instead of a Van Veen grab. Nevertheless, the shift remains difficult to interpret, highlighting the importance of maintaining consistent sampling methods in future monitoring to reduce uncertainty and improve comparability across years.

The 2026 results indicate a decrease in abundance but an increase in biomass relative to 2025, suggesting that the community may have shifted towards fewer but larger-bodied individuals. This pattern is broadly consistent with the increase in biomass relative to abundance observed at some of the other monitoring areas, although the ecological significance of this pattern at Elandsfontein remains uncertain due to the historical change in sampling equipment. Additionally, the temporal variability seen in 2026 remained within the established range of historical abundances and biomass records.

Results from multivariate analyses indicated considerable overlap in benthic infaunal communities in samples collected over the years at Elandsfontein, with only minor year-specific groupings (Figure 8.22). Overall, however, benthic infaunal community structure has remained relatively stable over time, with fluctuations in abundance and biomass more strongly linked to environmental and habitat differences than to long-term shifts in benthic assemblages.

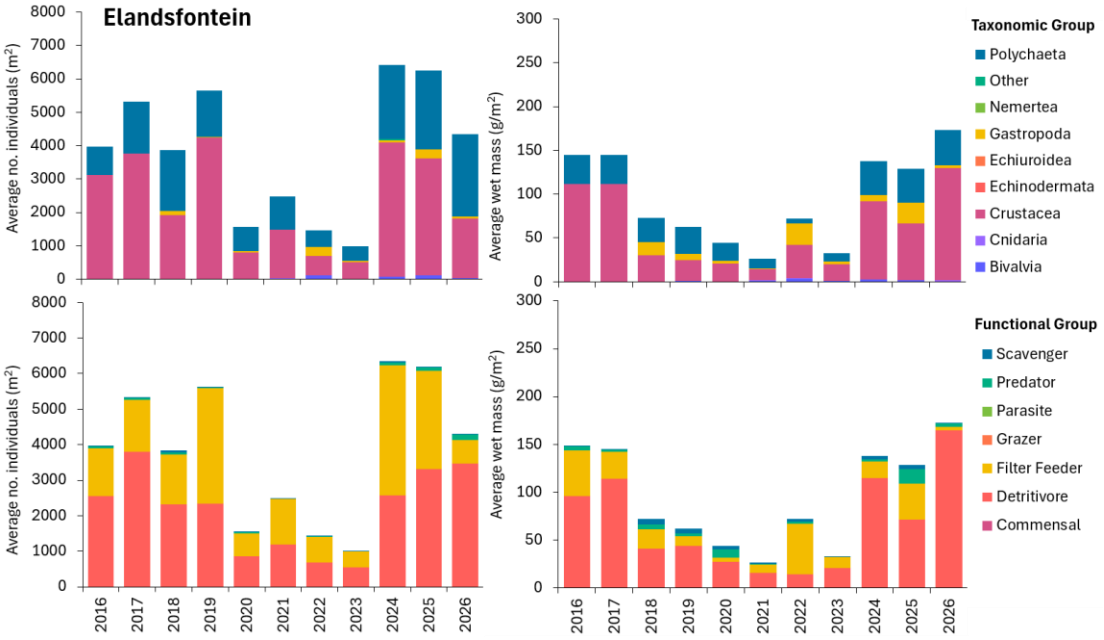


Figure 8.21. Overall trends in the abundance and biomass (g/m^2) of benthic infauna at Elandsfontein as shown by taxonomic and functional groups.

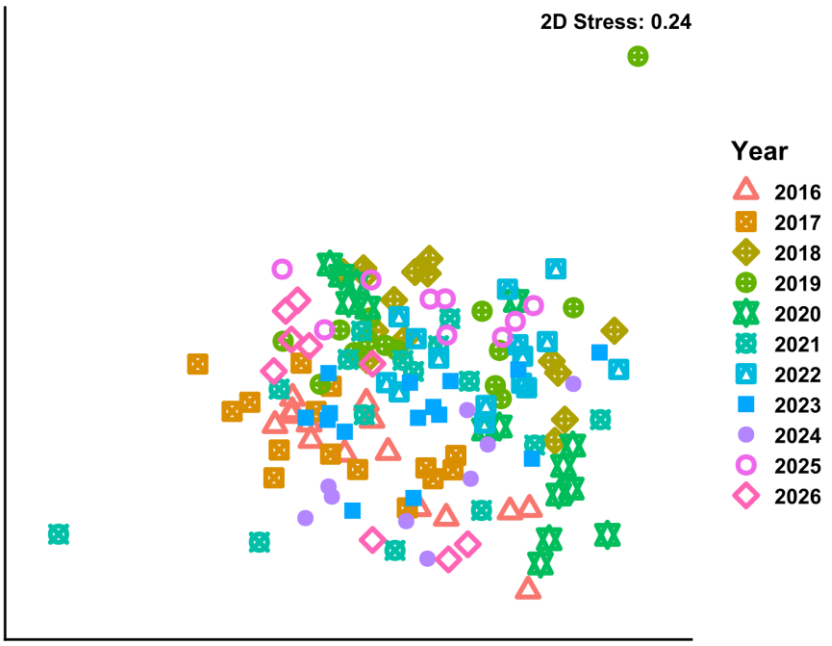


Figure 8.22. nMDS plot based on infaunal abundance data from samples collected at Elandsfontein from 2016–2026.

8.8 Discussion

Several studies have examined the infaunal community structure in Saldanha Bay, focusing on the impacts of human activities on benthic health. Substantial changes in these communities were observed before and after the harbour's development in the early 1970s, when approximately 25 million m^3 of sediment was dredged to build the harbour wall (Moldan 1978). This led to a decline in several species, particularly suspension feeders, while opportunistic scavengers and predators such as crabs and polychaetes increased (Moldan 1978, Kruger et al. 2005). One notable loss was the disappearance of the sea pen *Virgularia schultzei,* replaced by these opportunistic species (Moldan 1978). Kruger et al. (2005) similarly found that the benthic communities recorded prior to harbour development were significantly different from those observed in 2001, with the previously broad, continuous communities becoming distinctly separated into Small Bay and Big Bay communities.

Dredging can affect benthic communities through several mechanisms, including direct removal or burial of organisms, increased turbidity and suspended sediment concentrations, disturbance of sediment structure, and the mobilisation of organic matter and contaminants. Decomposition of accumulated organic material may also contribute to oxygen depletion and localised anoxic conditions (Moldan 1978). These effects are consistent with the declines in abundance, biomass and diversity observed at LPG since 2019, and the continued low diversity recorded at this site in 2026. The persistent low diversity at LPG may be associated with increased disturbance following installation of the Single Point Mooring, although other site-specific environmental factors may also contribute. Dredging activities, such as the 2010 Salamander Bay boatyard expansion, also had clear impacts on infaunal communities (Clark et al. 2011, 2012).

Historical observations provide further evidence of anthropogenic effects. Beckley

(1981) found that benthic communities near the IOT were dominated by pollution-tolerant polychaetes, while previous monitoring has documented declines in filter-feeding taxa during periods of dredging, including dredging events in 1996, 2007/2008 and 2009/2010. The sensitivity of suspension- and filter-feeding taxa to increased suspended sediment and altered sediment conditions provides a plausible explanation for their reduced abundance during periods of disturbance. In the current monitoring data, filter feeders are largely represented by *Upogebia capensis*, *Ampelisca* spp. and *Virgularia schultzei*. In contrast, detritivores have generally become more prominent during periods of disturbance. In Small Bay, this functional group is largely represented by *Listriolobus capensis* and polychaetes of the genera *Polydora* and *Mediomastus*.

Small Bay, in particular, remains subject to a combination of pressures that are likely to reinforce one another. Its sheltered position and altered hydrodynamics reduce water exchange and promote the deposition and retention of fine sediments, while inputs from fish-processing activities and mussel culture contribute organic material, further impacting and impoverishing benthic communities in parts of Small Bay. Of particular concern is the continued absence of *V. schultzei* from Small Bay, where the species was previously abundant, while it persists in Big Bay at low abundance. Given the documented changes in hydrodynamics and sediment characteristics within Small Bay, together with the elevated mud and TOC concentrations observed in this area, the loss of *V. schultzei* may reflect reduced habitat suitability. Nevertheless, long-term monitoring trends indicate increases in abundance and biomass at both Small Bay and Big Bay since monitoring began in the late 1990s. These increases may indicate some recovery of benthic communities following major dredging events; however, the continued absence of sensitive taxa at Small Bay and the prevalence of opportunistic species suggest that recovery may be incomplete.

Langebaan Lagoon has also been influenced by the arrival and establishment of the European black mussel, *Mytilus galloprovincialis*, which has been shown to have an influence on local benthic community structure (Hanekom & Nel 2002, Robinson & Griffiths 2002, Robinson et al. 2007). However, temporal trends indicate increase in species richness, abundance and biomass, suggesting recovery of the benthic communities.

Overall, the temporal trends observed do not indicate a system-wide improvement in benthic condition since dredging, with the Lagoon showing positive trends in all metrices (species richness, abundance and biomass). However, the strongest evidence of ongoing anthropogenic influence occurring in specific areas such as Small Bay and around the IOT. The continued presence of distinct benthic communities between Small Bay and Big Bay, together with the significant relationships between community composition and sediment-related variables, suggests that the physical habitat changes associated with harbour development continue to influence benthic community structure decades after construction.

For Elandsfontein, continued monitoring is particularly important given its naturally dynamic intertidal mudflat environment and the potential for future changes in groundwater quality and freshwater inputs associated with the proposed phosphate mine. Establishing a sufficiently long and consistent baseline will be essential for quantifying natural spatial and temporal variability and distinguishing potential future anthropogenic effects from background variability in the Elandsfontein–Langebaan Lagoon system.

9 FISH AND ECONOMICALLY IMPORTANT INVERTEBRATES

9.1 INTRODUCTION

The waters of Saldanha Bay and Langebaan Lagoon support an abundant and diverse fish community. 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*. Commercial netfishing continues in the area today, and although beach seines are no longer used, gillnet permit holders continue to target harders. Species such as white stumpnose, white steenbras, silver kob *Argyrosomus inodorus*, elf *Pomatomus saltatrix*, steentjie *Spondyliosoma emarginatum*, yellowtail *Seriola lalandi* and smooth hound shark *Mustelus mustelus* support large shore angling, recreational and commercial boat linefisheries 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 refuge from the cold, rough seas of South Africa's West Coast. The Saldanha-Langebaan Lagoon system is therefore considered a productive nursery area for early life stages of numerous fish species that are integral to ecosystem functioning (Clark et al. 2022).

The importance and long history of fisheries in the system has led to extensive scientific research on the fish resources and fisheries in the area, and includes archaeological studies of the middens surrounding Langebaan Lagoon, gillnet sampling to quantify bycatch in the commercial and illegal gillnet fishery, and a once-off rotenone survey for small cryptic fish species (Awad et al. 2004).

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 bluntnose guitarfish *Acroteriobatus blochii* (Næsje et al. 2008, Kerwath et al. 2009, Tunley et al. 2009, Attwood et al. 2010, Schultz 2010, Hedger et al. 2010, da Silva et al. 2013, 2021). Key findings of these studies include evidence that the Langebaan Lagoon Marine Protected Area (MPA) provides some protection for white stumpnose, particularly during the summer months that coincide 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 up to two 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 also showed long-term movements out of the Lagoon into the Bay and one individual was recaptured in Durban, confirming that long distance migration does take place for this species (Hedger et al. 2010). However, the fact that nearly all fish within the Bay were resident for one to two years after tagging, and the presence of young of the year juveniles in the surf zone, suggests that the elf population

within Saldanha Bay exhibits both migratory and resident spawning components (Hedger et al. 2010). Smooth hound sharks acoustically tagged within Langebaan Lagoon resided inside the Lagoon for more than a year, and two did not leave the MPA at all (da Silva et al. 2013). Tag-recapture data confirmed the site fidelity of smooth hound sharks to Langebaan Lagoon, with almost all recaptured sharks reported within the Lagoon, and some sharks recaptured after seven years at liberty (Klein et al. 2022). Additionally, a study of the reproductive biology, diet, and growth of smooth hound sharks in Langebaan Lagoon concluded that the Lagoon represents a pupping, nursery and feeding area for the largest and oldest hound sharks globally, emphasising the conservation importance of the MPA (da Silva et al. 2021). Overall, these studies have clearly demonstrated that these priority fishery species all derive protection from the Langebaan Lagoon MPA. Saldanha Bay and Langebaan Lagoon appears to function as a semi-closed system with respect to the demographics of many of the key fishery species (e.g., harders (Horton et al. 2019), white stumpnose (Kerwath et al. 2009, Attwood et al. 2010), steentjies (Tunley et al. 2009), elf (Hedger et al. 2010), and smooth hound sharks (da Silva et al. 2013)).

Given the importance for fish of the Saldanha-Langebaan system, ongoing long-term monitoring of fish populations in Saldanha Bay and Langebaan Lagoon is essential when assessing the status of many fish species in both a local and national context. Nationally, 31% of South Africa's fish stocks are depleted or heavily depleted, and many of the fish that utilise the Saldanha-Langebaan system to some degree (e.g., white stumpnose, blacktail *Diplodus capensis*, elf and harders) continue to be overfished and are overexploited (DFFE 2023, 2025). Furthermore, Saldanha Bay faces many anthropogenic pressures and needs to strike a balance between sustainable commercial and industrial development, while not compromising the ecological role this site plays in the maintaining and supporting marine species and habitats.

9.2 METHODS: EXPERIMENTAL FISH SURVEYS: 1994–PRESENT

Historical experimental beach seine net sampling of near-shore, sandy beach fish assemblages in Saldanha Bay and Langebaan Lagoon includes those conducted over short periods during 1986–1987 (University of Cape Town (UCT) Zoology Department, unpublished data), in 1994 (Clark 1997), and 2007 (Anchor Environmental Consultants and 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 carried out (Arendse 2011).

Experimental beach seine net sampling of near shore juvenile fish assemblages has been undertaken annually by Anchor since 2008, at 15–16 sites throughout the Saldanha-Langebaan system (Table 9.1 and Figure 9.2). The number, location and area sampled differed slightly between each of the surveys, but over the life of the monitoring programme, the sampling methodology has remained consistent.

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 2026 seine net survey (the 22nd year of sampling) and investigates trends in the fish communities by comparing this with data from previous seine net surveys (1994, 2005, 2007–2025).

Table 9.1. Experimental beach seine surveys in Saldanha Bay and Langebaan Lagoon, which contributed data to this study.

Period	Area sampled	No. hauls	No. sites
1994	Bay & Lagoon	39	8
2005	Bay & Lagoon	33	11
2007	Bay & Lagoon	21	7
2008–2012	Bay & Lagoon	36–42	15
2013–2026	Bay & Lagoon	48	16

Figure 9.1. Hauling in the beach seine net during an annual State of the Bay survey. Photo: J. Turpie.

Beach seine net surveys were conducted annually during late summer to early autumn (March–April), coinciding with peak recruitment of juvenile fish to the near-shore environment. Each year, 16 pre-determined sites were sampled using a standardised beach seine net, 30 m long and 2 m deep, with a stretched mesh size of 12 mm (Figure 9.1).

At each site (Figure 9.2) three replicate hauls were conducted approximately 50 m apart, during daylight hours. The net was deployed from a small inflatable boat 30–50 m from the shore and pulled to shore by hand. Weather conditions, tidal state, surf zone width and swell height were all estimated and recorded at each site. All fish captured were identified to species level (where possible; larval fish to family level), counted and a measured (Total Length) (up to 100 individual fish per species). For species that were too difficult to identify in the field, specimens were retained for post survey identification. Large fish hauls were sub-sampled on site, the size of the sub-sample estimated visually, and the remainder of the catch released alive.

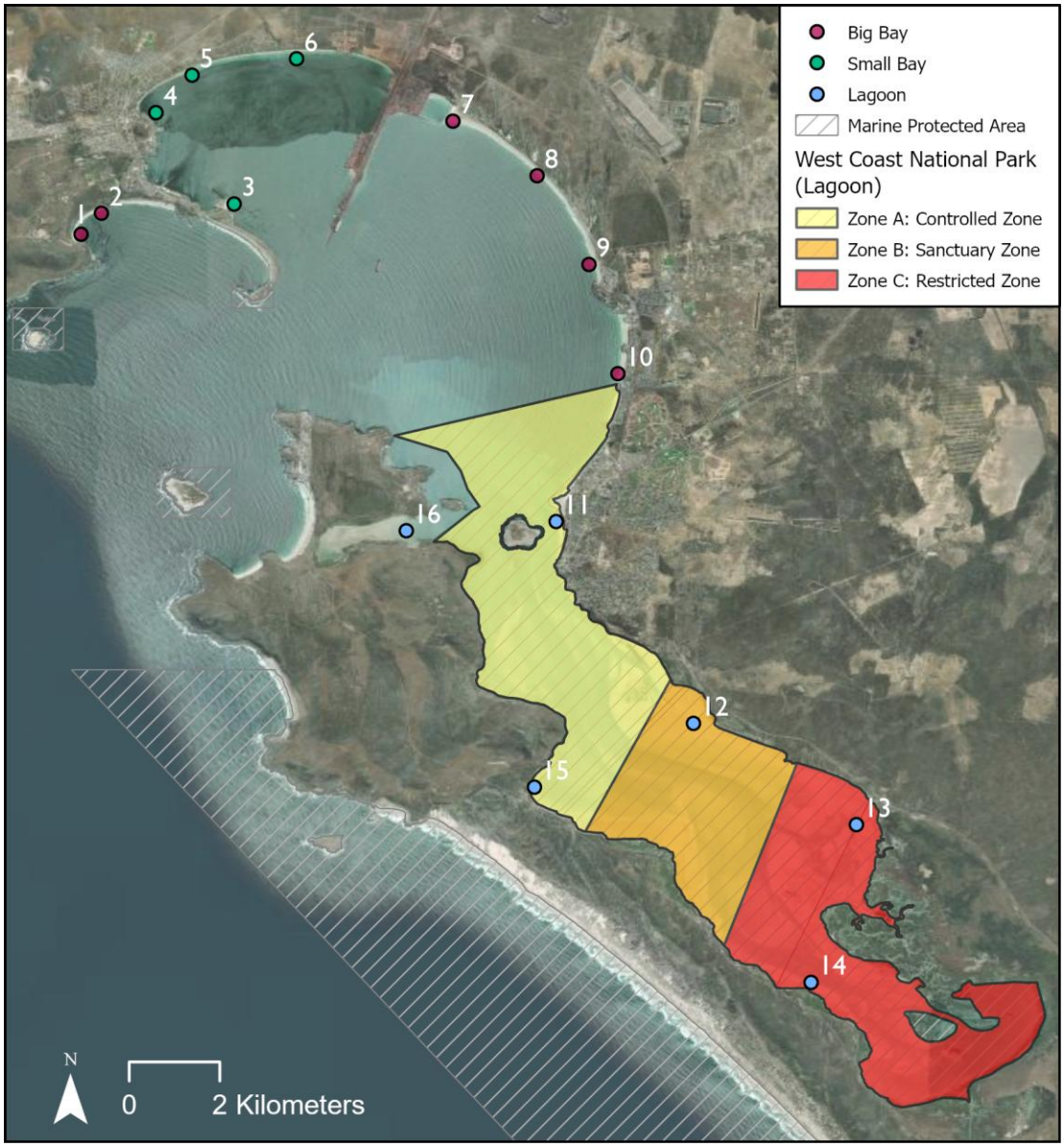


Figure 9.2. 2013–2026 beach seine net sampling sites within Saldanha Bay and Langebaan Lagoon, separated by region and in the context of the West Coast National Park (Langebaan Lagoon) MPA zones. 1: North Bay west, 2: North Bay east, 3: Small Craft Harbour, 4: Hoedjiesbaai, 5: Caravan Site, 6: Blue Water Bay, 7: Seafarm Dam, 8: Spreeuwalle, 9: Lynch Point, 10: Strandloper, 11: Schaapen Island, 12: Klein Oesterwal, 13: Bottelary, 14: Churchhaven, 15: Kraalbaai, 16: Rietbaai.

9.2.1 SPECIES IDENTIFICATION

Due to concerns around the accuracy of identification of certain species, *Caffrogobius* spp. gobies and *Clinus* spp. klipvis were grouped under their respective genera. Early SOB reports identified the guitarfish (also known as sandsharks) present in the system as lesser guitarfish *Acroteriobatus annulatus*, but the literature suggests that these were likely to have been bluntnose guitarfish *A. blochii*, as these species are frequently misidentified and *A. blochii* is much more likely to occur on the West Coast (Attwood & Mann 2013, Pollom et al. 2020, Ebert et al. 2021, van Staden et al. 2022). Consequently, all records of *A. annulatus* were considered as misidentifications and reclassified as *A. blochii* for this report. Furthermore, it is likely that the 2010 and 2012 identifications of

bluefin gurnard *Chelidonichthys kumu* were actually Cape gurnard *C. capensis*, which has occurred in every survey, and as a result, *C. kumu* was classified as *C. capensis* for this report. Recent genetic studies indicate that *Haploblepharus* spp. shysharks are also likely to have been misidentified in the past, as these species have been determined to be highly morphologically similar and possibly prone to hybridisation (van Staden et al. 2020, 2023). In response to this, *Haploblepharus* spp. were only identified to genus level.

9.2.2 Data analysis

The number of fish caught was corrected for any sub-sampling of large hauls that took place in the field prior to data analysis. The frequency with which each species or genus occurred across the 22 surveys was calculated.

The number of species caught (species richness) and average abundance of fish (all species combined) during each survey was calculated. Abundance was expressed as the number of fish per haul. For spatial comparisons, the system was divided into three regions: Small Bay, Big Bay and Langebaan Lagoon (see Figure 9.2). For the purpose of visualising differences in juvenile fish communities between regions over time, the average number of fish per haul, per species, was categorised and plotted for each region. Any species which only occurred once throughout the entire dataset was excluded from this analysis. When comparing the differences in overall abundance per region, data above the 95th percentile were omitted, to account for the skewing effect of some exceptionally large hauls of harders and Cape silversides.

To further investigate temporal variation in recruitment of species important in the Bay's fisheries (blacktail, elf, harders, bluntnose guitarshark and white stumpnose), the trends in total abundance of these species, across all regions, was assessed. These temporal trends were analysed using a Kruskal-Wallis test and, in the case of significant results, a post-hoc Dunn's test using the Holm method, in R (R Core Team 2022) and RStudio (Posit team 2025). The trends in average abundance of these species over time were plotted, including standard errors.

9.3 Results and discussion

9.3.1 Interannual trends in fish species diversity & communities

The total species richness (includes species grouped for analysis) in all seine surveys to date remained at 56 species, from 33 families, with the most recent addition being stompkop klipvis *Clinus cottoides*, which were found at North Bay (Big Bay) in 2025. Lumping of some species groups reduces this count to 48 taxa (see Section 0). Total species richness across all surveys remained highest in Big Bay (42) intermediate in Small Bay (31), and lowest in the Lagoon (25). Average species richness was similar between Small Bay (12.0) and Big Bay (12.6), and on average lower in the Lagoon (10.2) (Figure 9.3). Slightly more variation in the number of species caught over the period of sampling was apparent for Big Bay, where the most diverse samples were collected in 2012 (21 species) including the only occurrences of sardine *Sardinops sagax*, John Dory *Zeus faber*, white seacatfish *Galeichthys feliceps*, and angelfish *Brama brama* in the history of the surveys.

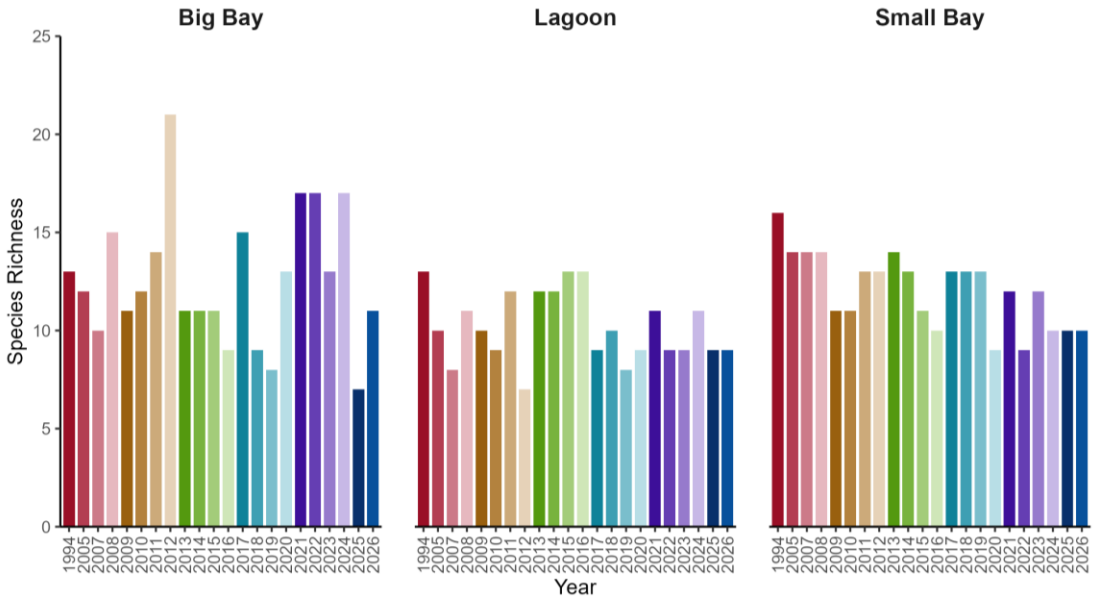


Figure 9.3. Species richness (seine net surveys) in each area of Saldanha Bay and Langebaan Lagoon, over the period 1994–2026.

Fish species richness in 2026 was again lower than average in Small Bay and the Lagoon, at 10 species and nine species, respectively (Figure 9.3). In 2026, 11 species were caught in Big Bay, an improvement over the seven species caught in 2025 (Figure 9.4). Small Bay showed a concerning decline in species richness over time (Figure 9.3).

The species composition in the different areas changed substantially between years, but the same ubiquitous species were recorded in nearly all surveys. Two species, Cape sole, which is classified as Near Threatened on the IUCN Red List, and harder occurred every year in every region, while Cape silversides, *Caffrogobius* spp. gobies, Cape gurnard, and klipvis *Clinus* spp. occurred in every year but not necessarily in every region each year (Table 9.2). These species make up key components of the Saldanha Bay-Langebaan Lagoon juvenile fish community (Figure 9.5, Figure 9.4, Figure 9.6).

Other frequently occurring species which have occurred for at least 22 years include bluntnose guitarshark and white stumpnose, which is classified as Vulnerable on the IUCN Red List (IUCN 2025) (Table 9.2). White stumpnose has occurred in every region in every year, except 2023 when it was not recorded in any region. Bluntnose guitarshark was not recorded in 2025 for the first time in 22 years of sampling.

Table 9.2. All bony fish and chondrichthyan taxa caught during in 22 beach seine net surveys of Saldanha Bay and Langebaan Lagoon conducted over the period 1994–2026. The IUCN Red List global status of threatened species is indicated (NT = Near Threatened, VU = Vulnerable, EN = Endangered, CR = Critically Endangered) (IUCN 2025), as well the % occurrence of each taxon in annual surveys over the sampling period.

Family	Species/taxon	Common name	Occurrence in 22 surveys
Teleosts (bony fish)			
Alosidae	*Sardinops sagax*	sardine/pilchard	5%
Ariidae	*Galeichthys feliceps*	white seacatfish	5%

Family	Species/taxon	Common name	Occurrence in 22 surveys
Atherinidae	*Atherina breviceps*	Cape silverside	100%
Blenniidae	*Parablennius cornutus*	horned blenny	27%
	Parablennius pilicornis	ringneck blenny	5%
Bramidae	*Brama brama*	angelfish/pomfret	5%
Carangidae	*Lichia amia*	leervis/garrick	5%
	Trachurus trachurus[VU]	Atlantic horse mackerel	36%
Centrolophidae	*Hyperoglyphe antarctica*	Antarctic butterfish	5%
Clinidae	*Blennophis* spp.	*Blennophis* klipvis spp.	9%
	Cancelloxus longior	Elongate sand klipvis	82%
	Clinus spp.	klipvis spp., including *Clinus agilis*, *C. cottoides*, *C. heterodon*, *C. latipennis*[EN], *C. robustus, C. superciliosus*, *C. venustris*, and unidentifiable *Clinus* sp.	100%
Cynoglossidae	*Cynoglossus capensis*	Sand tonguefish	5%
Dichistiidae	*Dichistius capensis*	galjoen	9%
Dussumieriidae	*Etrumeus whiteheadi*	red eye round herring	5%
Ehiravidae	*Gilchristella aestuaria*	estuarine round herring	5%
Engraulidae	*Engraulis japonicus*	Cape anchovy	5%
Gempylidae	*Thyrsites atun*	snoek	5%
Gobiesocidae	*Chorisochismus dentex*	rocksucker	9%
Gobiidae	*Caffrogobius* spp.	goby spp., including *Caffrogobius caffer*, *C. gilchristi*, and *C. nudiceps*.	100%
	Psammogobius knysnaensis	Knysna sandgoby	95%
Gonorynchidae	*Gonorynchus gonorynchus*	beaked sand eel	45%
Mugilidae	*Chelon richardsonii*	harder	100%
Parascorpididae	*Parascorpis typus*	jut jaw	5%
Pomatomidae	*Pomatomus saltatrix*	elf	82%
Soleidae	*Heteromycteris capensis*[NT]	Cape sole	100%
	Pegusa nasuta	snouted sole	23%
	Solea turbynei	blackhand sole	27%
Sparidae	*Argyrozona argyrozona*[NT]	carpenter	18%
	Diplodus capensis	blacktail/dassie	86%
	Lithognathus sp.	steenbras sp.	27%
	Pterogymnus laniarius	panga	5%
	Rhabdosargus globiceps[VU]	white stumpnose	95%
	Rhabdosargus holubi	Cape stumpnose	18%
	Sarpa salpa	strepie	18%
	Spondyliosoma emarginatum	steentjie	45%
Syngnathidae	*Syngnathus temminckii*	longsnout pipefish	95%
Tetraodontidae	*Amblyrhynchote honckenii*	evileye blaasop	23%
Triglidae	*Chelidonichthys capensis*	Cape gurnard	100%

Family	Species/taxon	Common name	Occurrence in 22 surveys
Zeidae	*Zeus faber*	John Dory	5%
Chondrichthyans			
Callorhinchidae	*Callorhinchus capensis*	St Joseph's shark	14%
Dasyatidae	*Dasyatis chrysonota*[NT]	blue stingray	14%
Myliobatidae	*Myliobatis aquila*[CR]	common eagle ray	32%
Rajidae	*Raja clavata*[NT]	thornback skate	5%
Rhinobatidae	*Acroteriobatus blochii*	bluntnose guitarfish	95%
Scyliorhinidae	*Haploblepharus* spp.	shyshark spp.	32%
	Poroderma africanum	striped catshark	9%
Triakidae	*Mustelus mustelus*[EN]	common smooth hound shark	18%

In terms of the different regions, the juvenile fish community of Big Bay consistently comprises of these commonly occurring species (Figure 9.4). Cape gurnard has been recorded every year in Big Bay and has occurred much less frequently and in lower numbers in other areas of the system. Elf occurred most frequently and most abundantly in Big Bay and was recorded in 18 surveys. Elf were only absent in one of the 13 surveys conducted over the period 1994–2017 but were not recorded in Big Bay during the 2018–2021 surveys. While only one elf was caught in the 2022 survey, 59 elf were caught in the 2023 survey, 64 in 2025, and 137 in 2026, making it the 4th most abundant species in Big Bay in 2025 and 2026. The elongate sand klipvis *Cancelloxus longior* also occurred much more frequently in Big Bay than in the other regions, and was recorded in 18 surveys, albeit generally in low numbers (Figure 9.4).

Big Bay had the highest number of species which have only occurred once or twice in the survey's history (22 species), most likely due to the region's high connectivity with the open ocean. Furthermore, all species for which only one individual was ever recorded have occurred in Big Bay. These include John Dory, snoek *Thyrsites atun*, sardine, panga *Pterogymnus laniarius*, jut jaw *Parascorpis typus*, ringneck blenny *Parablennius pilicornis*, Antarctic butterfish *Hyperoglyphe antarctica*, white seacatfish, sand tonguefish *Cynoglossus capensis* and angelfish, which were all excluded from the heatmap (Figure 9.4).

Within Small Bay, Cape silverside, genus *Caffrogobius* gobies, harders, and Cape sole occurred in all surveys, while *Clinus* spp. klipvis and white stumpnose occurred in all but one survey each (Figure 9.5). Small Bay has the highest proportion of "resident" species that frequently occur, whilst a larger proportion of the Big Bay ichthyofauna, although more diverse, occurs sporadically in catches.

Over the sampling record, Small Bay tended to have more frequent occurrences (81% of surveys) and substantially greater abundance of blacktail compared to the rest of the system although only a single blacktail was caught in the 2026 survey. The longsnout pipefish *Syngnathus temminckii* also occurred most frequently and abundantly in Small Bay and was usually associated with macro algae that accumulates in the Small Bay surfzone. The Near Threatened thornback skate *Raja clavata* has only occurred in Small Bay, when four individuals were sampled in 2007. *Haploblepharus* species of shyshark have been recorded in six surveys in Small Bay, in comparison to only twice in Big Bay and once in the Lagoon (Figure 9.5).

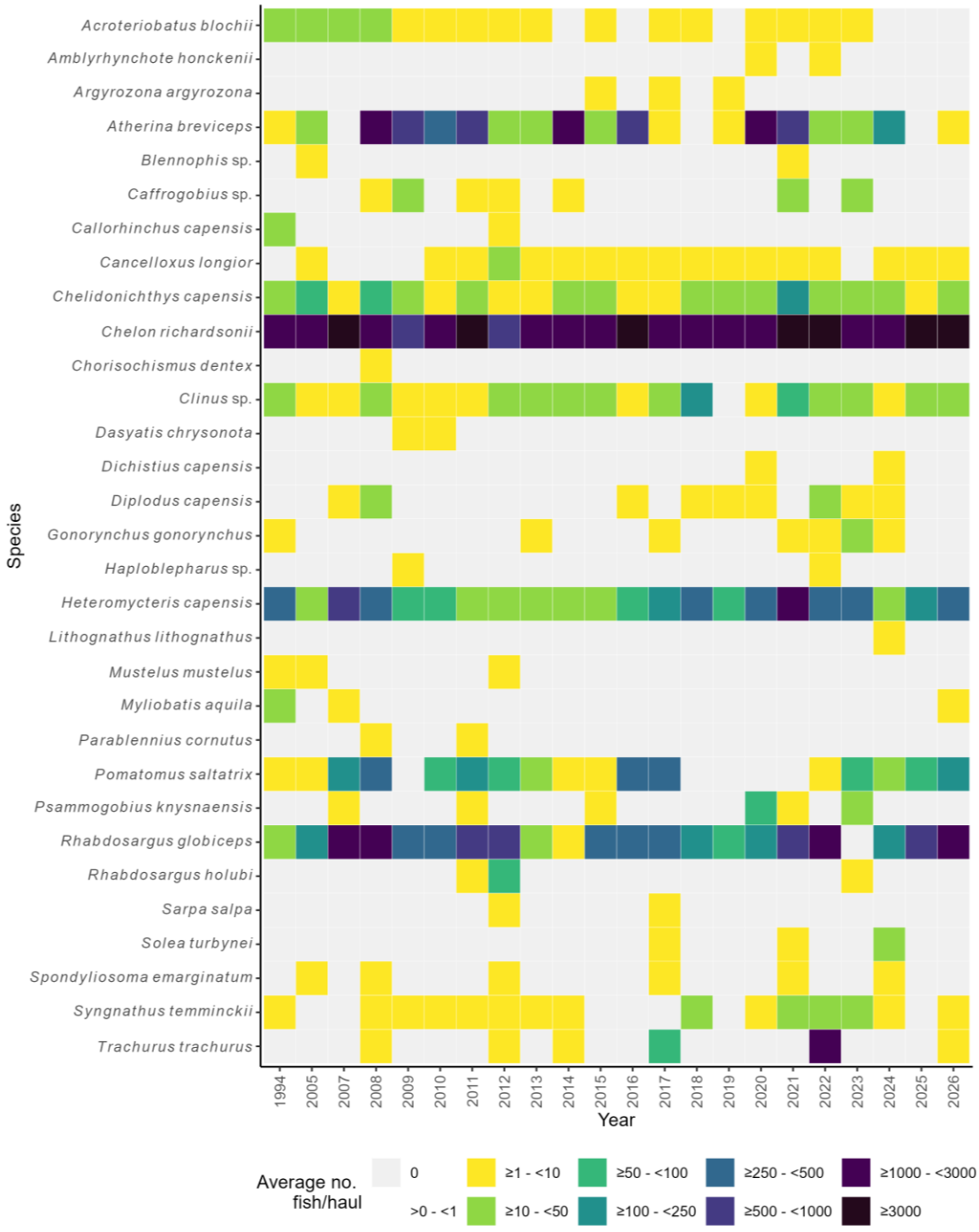


Figure 9.4. Juvenile fish species community (1994–2026) from beach seine surveys in Big Bay, Saldanha Bay, scaled by average number of fish caught per haul. Note that species occurring only once in the history of the surveys have been excluded.

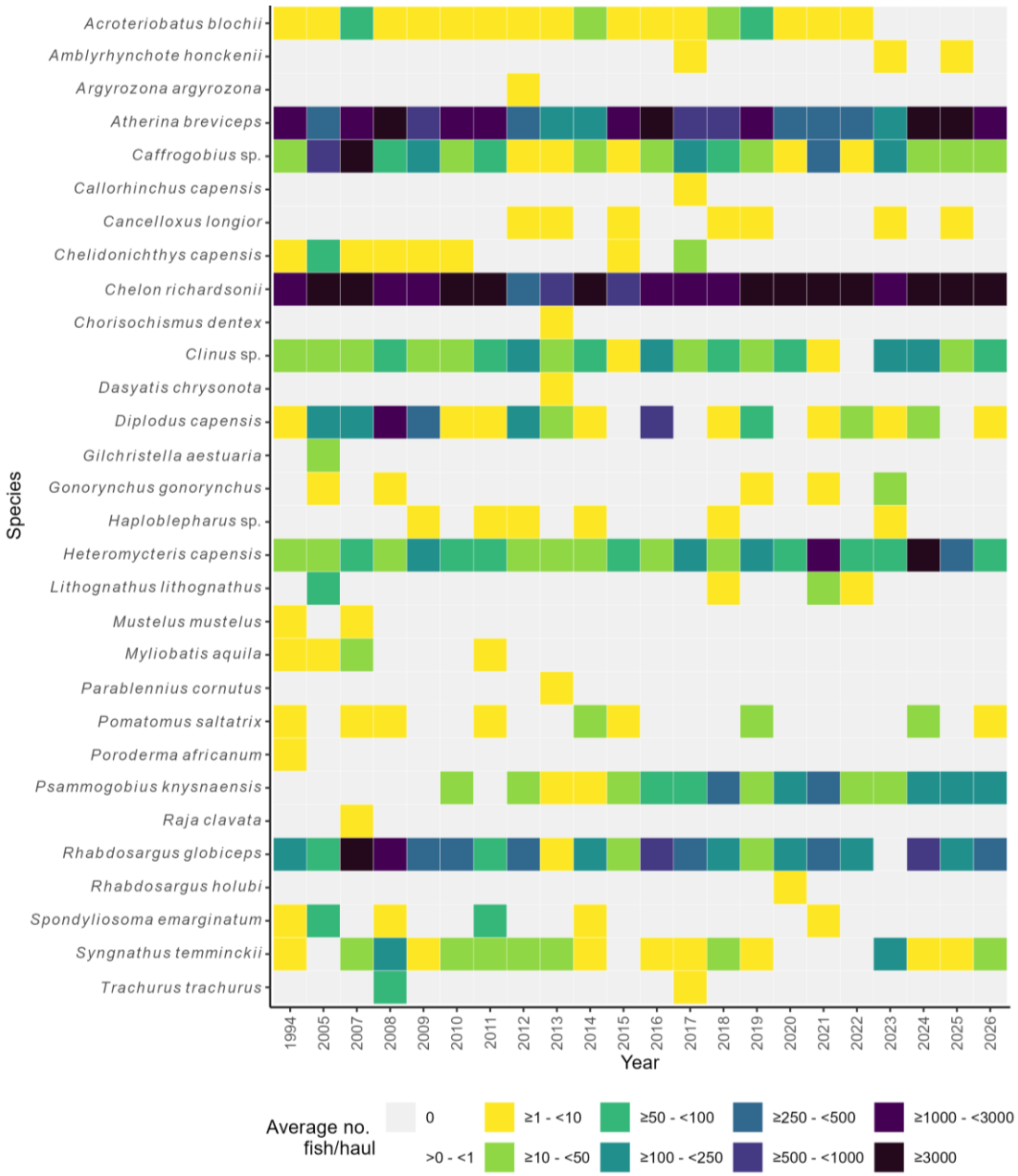


Figure 9.5. Juvenile fish species community (1994–2026) from beach seine surveys in Small Bay, Saldanha Bay, scaled by average number of fish caught per haul. Note that species occurring only once in the history of the surveys were excluded.

The fish community composition in the Lagoon is similar to that found in Small Bay in terms of the dominant species. Cape silversides, *Caffrogobius* spp. gobies, Cape sole and harders were all recorded in every survey, while Knysna sandgoby *Psammogobius knysnaensis* and white stumpnose occurred in all but one survey each (Figure 9.6). Knysna sandgobies and *Caffrogobius* spp. gobies were usually much more abundant in the Lagoon than in the other areas of the system. In 2025, the Lagoon surveys contributed 85% and 97% to the total Knysna sandgoby and *Caffrogobius* spp. catches, respectively. The snouted sole *Pegusa nasuta* only occurred in the Lagoon but was not recorded since 2011(Figure 9.6).

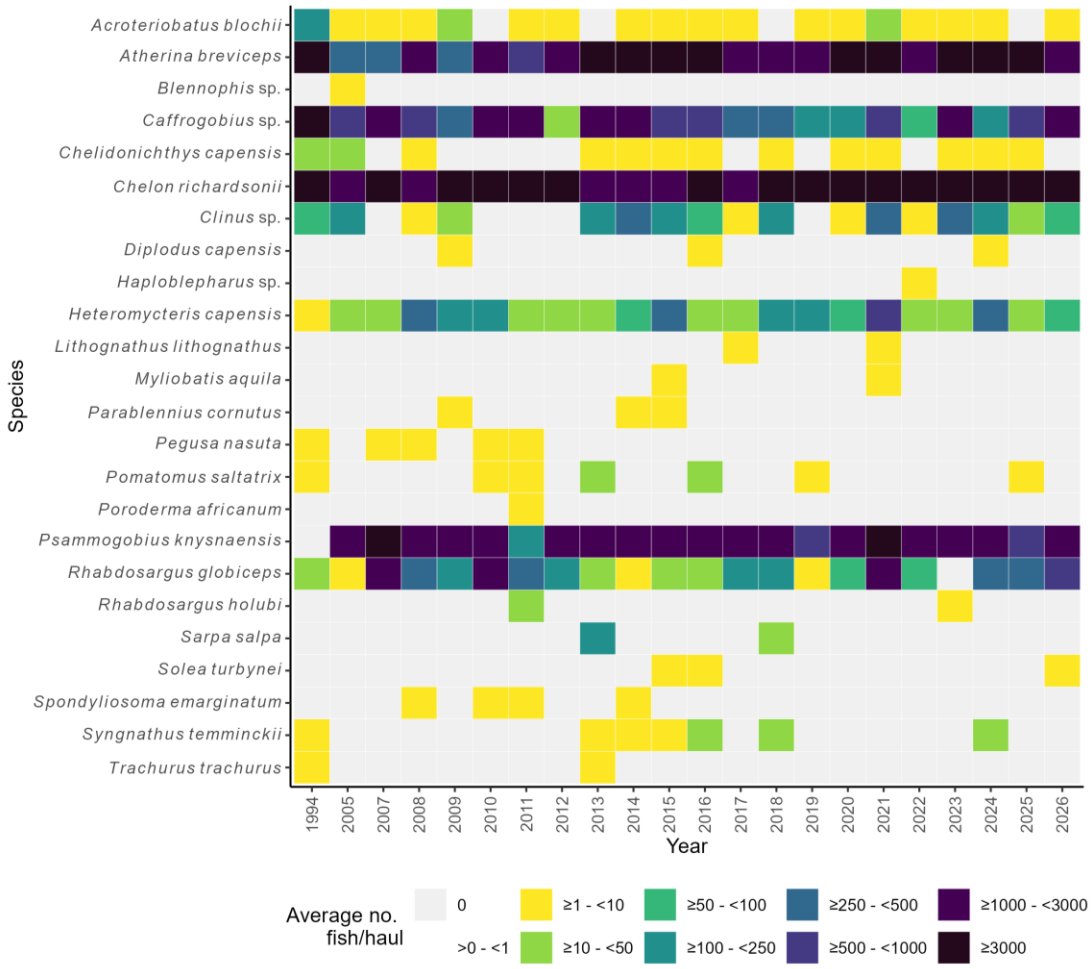


Figure 9.6. Juvenile fish species community (1994–2026) from beach seine surveys in Langebaan Lagoon, Saldanha Bay, scaled by average number of fish caught per haul. Note that species occurring only once in the history of the surveys were excluded.

9.3.2 Interannual Trends in Fish Abundance

The overall fish abundance showed high inter annual variability within, and between, the three areas of Saldanha Bay (Figure 9.7). There were no clear temporal trends in abundance. Harders and Cape silversides dominated the catches for nearly all surveys and large variation in the catches of these abundant shoaling species was the main cause of the observed variability between years (Figure 9.5, Figure 9.4, Figure 9.6).

In the Lagoon and Small Bay in particular, periodic large catches increased the inter annual variation. For example, in Small Bay in 2011, a total of 137 485 harders were caught, driven by a few very large catches when 133 943 harders were caught across three hauls at Caravan Site, leading to the abundance of fish in that year being highly variable between sites, and a lot higher on average than the succeeding year, 2012.

This large variability has a strong skewing effect on the long-term average abundance values, and as a result, the median value is a more appropriate measure. The overall median abundance (excluding the 95th percentile) in the Lagoon is highest at 410 fish/haul, followed by Small Bay at 295 fish/haul. The Big Bay long term median is lowest at 90 fish/haul (Figure 9.7).

The median fish abundance in the Lagoon in 2026 was similar to the long-term median for this area, and both the Small Bay and Big Bay 2026 medians were roughly double the long-term medians for these regions (Figure 9.7).

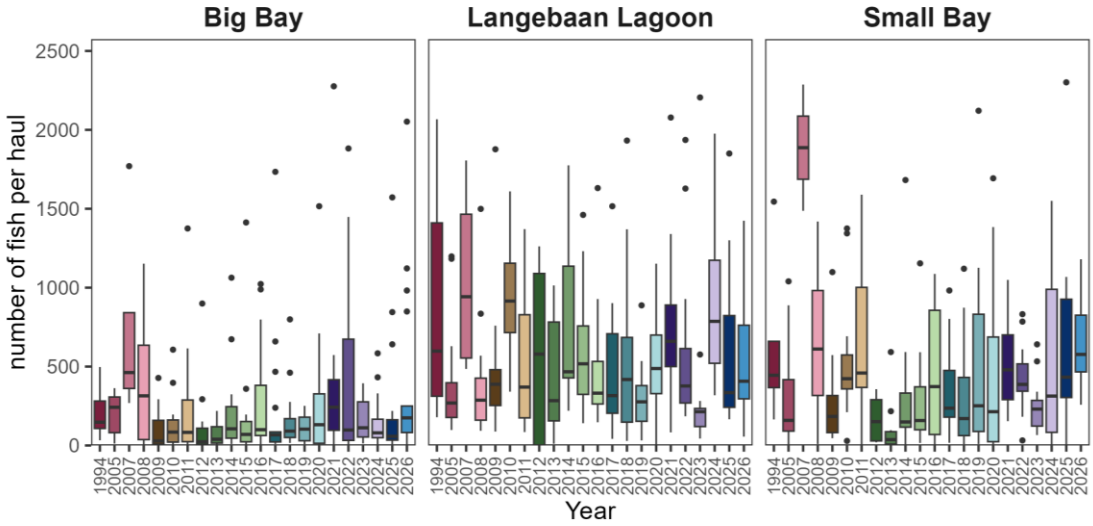


Figure 9.7. Number of fish per haul by region of Saldanha Bay and Langebaan Lagoon, 1994–2026. Note that data greater than the 95th percentile were omitted. Black dots indicate outliers. Boxes represent the range between the 25th and 75th percentile, divided by the median. Box whiskers represent the 25th percentile-1.5*the inter-quartile range and the 75th percentile+1.5*the interquartile range.

9.3.3 KEY FISHERY SPECIES

WHITE STUMPNOSE

White stumpnose *Rhabdosargus globiceps* is a medium sized, endemic sparid, and is recorded from southern Angola to the Kei River on the east coast of South Africa (Mann 2013, Whitfield & Mann 2023). It is an important recreational and commercial linefish species off the west and south coasts of South Africa in habitats of sandy sea beds, near small reefs and rocky outcrops (Griffiths et al. 2002). White stumpnose is the main angling target species within the Saldanha-Langebaan system. Recent studies on the national stock status shows that the stock is fully exploited or overexploited (DFFE 2023). Arendse (2011) used catch-at-age data from the Saldanha-Langebaan boat fishery and per-recruit modelling to estimate that spawner biomass in the Bay at the time (2006–2008) was less than 25% of pristine levels. 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 within the Saldanha-Langebaan system was required to achieve this target.

Since the peak in abundance in 2007, when an average of 1051 white stumpnose/haul were caught, there has been a notable decrease in numbers of juvenile white stumpnose throughout the system (Figure 9.8). It may be that the peak abundance in 2007 was the exception, and the lower abundance recorded before and after this period represent the

more typical situation. The concerning declining trend in white stumpnose was thought to have reversed with the third highest white stumpnose abundance recorded in 2021 samples, but it is not clear if this moderate "recovery" is being sustained. Abundance values from 2024, 2025 and 2026 were all significantly greater than the abundance in 2023 (Dunn's test, p <0.005). Although the average white stumpnose abundance between 2022 and 2026 was substantially greater than was observed between 2011 and 2020, longer-term data are required to have confidence in any positive trend. The average abundance of white stumpnose in 2026 was 44.8 fish/haul, which is 1.7 times the average abundance observed in 2025 of 26.9 fish/haul, and substantially greater than the long-term average abundance since 2008 (after the anomalous high catches in 2007), of 25.02 fish/haul. However, white stumpnose numbers are still concerningly low.

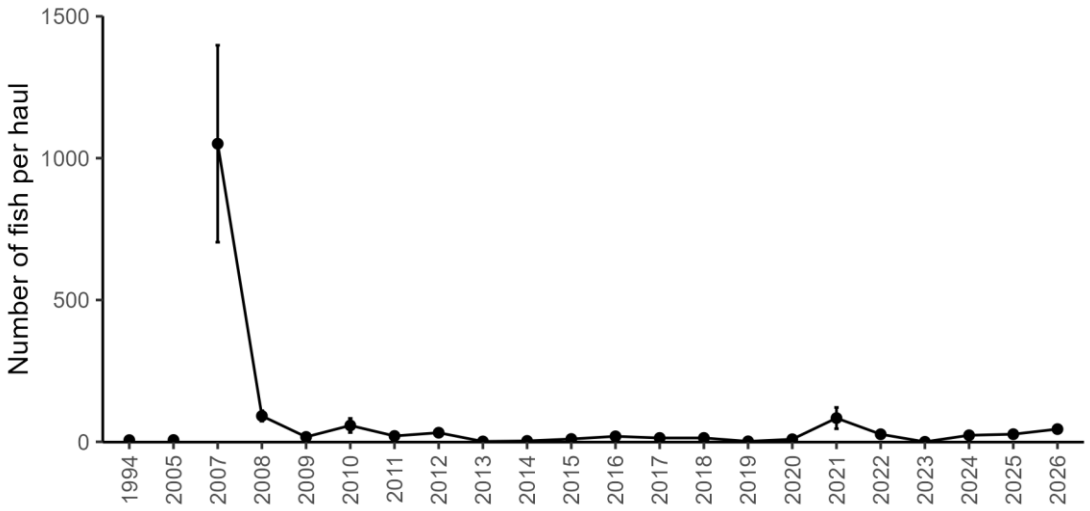


Figure 9.8. Average annual catch per haul of juvenile white stumpnose caught in beach seine net sampling in Saldanha Bay and Langebaan Lagoon (1994 and 2005–2026). Standard Errors are displayed.

Results from angler surveys undertaken in Langebaan during the early 2000s indicated that approximately 92 tonnes of white stumpnose were landed by anglers each year (Næsje et al. 2008). Parker et al. (2017) provided an updated analysis of angler survey data, commercial linefish catch returns and the juvenile white stumpnose catch in the annual seine net surveys between 2000–2016, which conclusively demonstrated substantial declines in both adult and juvenile abundance estimates over that time frame.

Increased fishing pressure over the last three decades, particularly by the recreational sector, has led to concerns regarding sustainability of the local white stumpnose stock. 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), driven by environmental variations amongst other factors, but given the sustained declines throughout the system, and the findings of Arendse (2011) and Parker et al. (2017), it appears that recruitment overfishing is the likely 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 does not have the reproductive capacity to replenish itself. It has been suggested that changes to

management measures are required to rebuild white stumpnose stock(Parker et al. 2017). Specifically, it was recommended 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) should be implemented (Arendse 2011). In 2025, Anchor assisted the WQT in drafting a letter to the DFFE to motivate for these management actions, but no changes to the management regulations have been implemented to date. The Langebaan Yacht Club however, has erected signs encouraging the many fishers who use their slipway to adhere to these guidelines.

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). These life history strategies (relatively rapid growth and early maturity) in combination with the protection afforded by the Langebaan Lagoon MPA are probably part of the reason that stocks of white stumpnose in Saldanha and Langebaan Lagoon had until recently been resilient to rapidly increasing recreational fishing pressure. However, the low abundances in the last decade suggest that the Langebaan Lagoon MPA alone may not be enough to prevent stock collapse, or to ensure recovery of the stock, at current levels of fishing effort (Arendse 2011, Parker et al. 2017). Furthermore, should any recovery happen in white stumpnose recruitment (the survival and growth of juvenile fish into adulthood), it is likely that under the current management measures (bag and size limits) this would be undermined by overfishing once the individuals reach adulthood and are large enough to be caught by fishers.

HARDER

The Southern mullet, or harder, *Chelon richardsonii* can be found in South African coastal waters, mainly off rocky points and sandy beaches, as well as in estuaries which serve both as adult habitat and nursery areas. They remain in the surf-zone and nearshore marine environment for most of the year and form large shoals in autumn and spring, often in response to food availability (Mann 2013). Harders are the main target species of the inshore net fishery (gillnet and beach-seine) on the west coast of South Africa and experience heavy fishing pressure (Horton et al. 2019, DFFE 2025). Located in the middle of the species' distribution, Saldanha Bay and Langebaan Lagoon provide essential nursery grounds and refugia for both juvenile and adult harders.

An investigation of the age, growth and stock assessment of the harder stock in the Saldanha-Langebaan system was completed in 2019, which showed that gillnet fishers have seen substantial declines in harder catches and Catch-Per-Unit-Effort (CPUE) over

the last two decades (Horton et al. 2019). By comparing monitored landings with reported catches, Hutchings & Lamberth (2002b) estimated that 400–600 tonnes of harders, valued at approximately R1.8 million at the time, was caught in the Saldanha-Langebaan system during 1998–1999. The reported catch has declined from around 130 tonnes per year over the period 2008–2012 to about 90 tonnes per year over the period 2013–2016, whilst effort remained fairly constant (Horton et al. 2019). The average size of harders in catches has declined significantly over the same period and a stock assessment indicated that the stock is heavily depleted and at risk of recruitment failure (the current spawner biomass was estimated at less than 24% of the pristine level) (Horton et al. 2019, DFFE 2025). A reduction in fishing mortality (approximately 30%) and an increase in mesh size to 51 mm was recommended to help rebuild the stock but has not been implemented to date (Horton et al. 2019). Furthermore, as illegal gillnetting appears to be having a significant negative effect on the harder population, it is likely that management of the legal fisheries is only part of the solution required (DFFE 2025).

Harders are the most abundant harvested species and inter annual variation in abundance of recruits was greater than for the other species, with average abundance in 2007, 2010 and 2011 being much greater than most other sampling events (Figure 9.9). The average abundance for these years had high standard error values, indicating that a few large hauls of harders were contributing to the intra- and interannual variability. Since 2011, the abundance of juvenile harders has been relatively low with less variation. However, there is some indication of a positive trend in harder numbers in recent years, and 2025 saw the greatest abundance of harders captured since the 2011 peak, mostly due to a haul of 12 700 harders at the Caravan Site (Small Bay), as well as a few other large hauls in Big Bay and the Lagoon, resulting in an overall average abundance of 687 harders/haul for 2025 (Figure 9.9). Abundance dropped slightly in 2026 to an average of 598 harders/haul, but this is still greater than the average of the last 15 years. Small Bay and Langebaan Lagoon are considered most important nursery areas for this species and have the greatest average abundance of juvenile harders over the long-term and in the last five years.

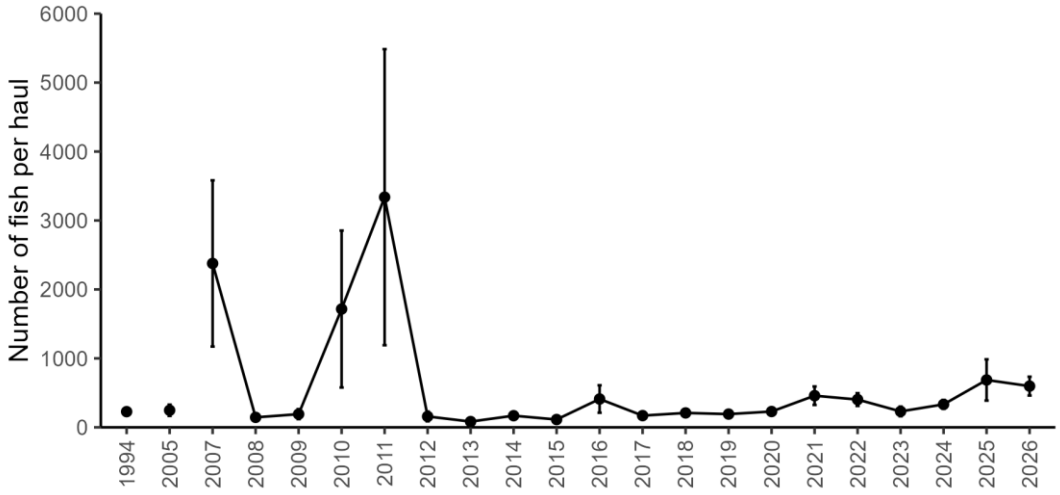


Figure 9.9. Average annual catch per haul of juvenile harders caught in beach seine net sampling in Saldanha Bay and Langebaan Lagoon (1994 and 2005–2026). Standard Errors are displayed.

BLACKTAIL

Blacktail *Diplodus capensis* is a very common inshore coastal species endemic to southern Africa that has two separate populations, one along the coast of South Africa from Saldanha Bay to southern Mozambique and the other from southern Namibia to southern Angola (Mann 2013). They are highly resident and rarely leave their preferred home region (Mann 2013, Watt-Pringle et al. 2013). Juveniles are typically found in intertidal rockpools, shallow subtidal reefs and gullies, sandy beach surf-zones and estuary mouths (Whitfield & Mann 2023). Blacktail are heavily exploited in the recreational, small-scale and subsistence fisheries, and the population is considered depleted and overexploited at a national level (DFFE 2023). It is a very popular angling fish along the entire South African coastline and within Saldanha Bay.

Beach seine sampling data show that blacktail abundance peaked in 2008, at an average of 34 fish/haul, but since then there has been a scarcity or absence of blacktail recruits (Figure 9.10). Although small peaks in abundance were observed subsequently, in 2012 and 2016, the annual surveys since 2008 have only sampled an average of 1.6 fish/haul (Figure 9.10). Since the 2016 peak in abundance, the average blacktail catch has been substantially lower than the long-term average, at 0.3 fish/haul. Last year (2025) was the third year in the survey's history that zero blacktail were caught, and in 2026 only a single individual was caught, in Small Bay (Figure 9.10).

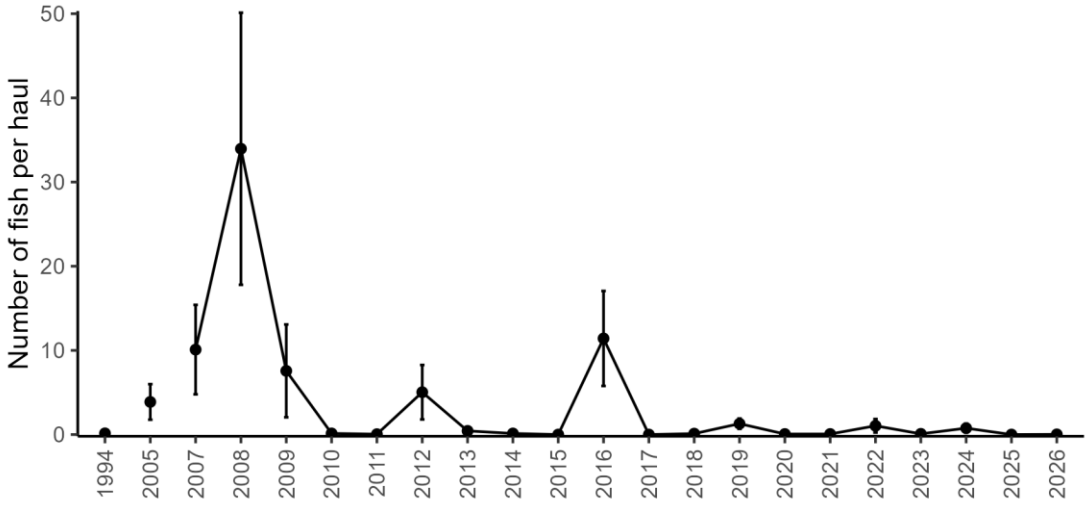


Figure 9.10. Average annual catch per haul of juvenile blacktail caught in beach seine net sampling in Saldanha Bay and Langebaan Lagoon (1994 and 2005–2026). Standard Errors are displayed.

Similarly to white stumpnose, this fish has been overexploited both locally and nationally and a reduction in fishing mortality is likely required to rebuild blacktail stocks. As mentioned in Section 9.3.1, blacktail juveniles have a strong preference for Small Bay compared to Big Bay and the Lagoon, and almost 100% of blacktail catches in the seine net surveys occur in Small Bay. This means that trends in the abundance of blacktail in the seine net survey data generally reflect the status of blacktail in Small Bay. The highly resident behaviour of blacktail renders these fish vulnerable to localised environmental

impacts such as pollution, as they are less likely to move out of areas that are negatively affecting them (Watt-Pringle et al. 2013). Consequently, the health of Small Bay is critical to the Saldanha Bay blacktail population. On a more positive note, divers collecting biofouling samples from the wreck near Klein Oesterwal reported a large shoal of adult blacktail present, so spawner biomass may be protected within Zone B of the MPA. However, it is equally important that the main blacktail nursery habitat in Small Bay remains healthy for this population to persist.

ELF

Elf *Pomatomus saltatrix* (also known as shad in South Africa, bluefish in North America and Europe, and tailor in Australia) occurs along the entire southern African coast but is less common between Cape Point and northern Namibia, except within Saldanha Bay, St Helena Bay and the three large, open estuaries along the west coast (Berg, Olifants and Orange). They are a pelagic, shoaling species found over sandy and rocky substrata from the surf-zone down to 100 m. Juveniles prefer shallow subtidal embayments, as suggested above, mainly along the south-eastern and south-western Cape coast (Maggs & Mann 2013). Despite high inter-annual variability, this species is considered to be heavily depleted and overexploited in South African waters with the spawner biomass estimated to be less than 40% of its pristine level (DFFE 2023). They are listed as Vulnerable on the IUCN Red List (Carpenter et al. 2015). Elf is the most important species caught by recreational shore anglers in South Africa, with fair numbers caught by the beach seine fishery in the Western Cape and as bycatch in the gill net fisheries, with overall heavy fishing pressure (Hutchings & Lamberth 2002a, Maggs & Mann 2013, DFFE 2023). Acoustic telemetry shows evidence of elf residency in the Langebaan Lagoon MPA which likely plays an important role in protecting both adults and juveniles of this species (Hedger et al. 2010).

Juvenile elf abundance has shown high variation in the last 20 years of sampling, with strong recruitment in Big Bay evident in the 2007,2008, 2016 and 2017 sampling that has not re-occurred (Figure 9.11).

Elf start to become sexually mature at one year (Maggs & Mann 2013), but as larger and older fish spawn exponentially more eggs, these strong cohorts need to survive multiple years beyond maturity to be able to contribute significantly to recruitment in the Bay (presuming the cohort does not emigrate from the Bay). The concerning scarcity of elf recruits since 2018 may be a result of reduced spawner biomass due to high fishing mortality or environmental factors (almost certainly both). The 2023 to 2026 catches of elf were greater than what was observed between 2020 and 2022, when only one elf was caught across three years, and show a positive trend, suggesting a slight increase in elf recruitment (Figure 9.11). Indeed, the 2026 average abundance of 2.9 fish/haul was the greatest observed since the peak in 2017. Future years of monitoring of elf will reveal whether this recent slight improvement in recruitment continues.

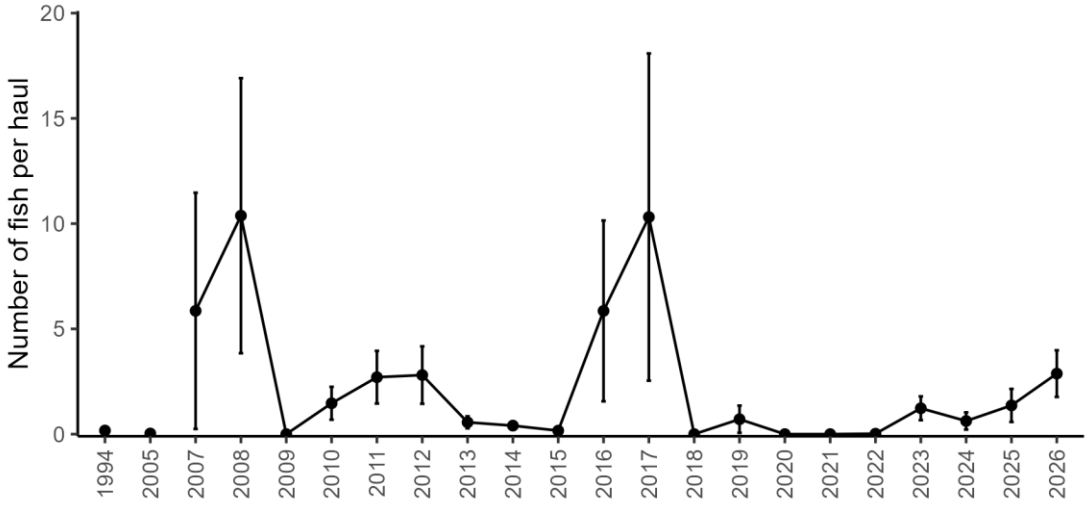


Figure 9.11. Average annual catch per haul of juvenile elf caught in beach seine net sampling in Saldanha Bay and Langebaan Lagoon (1994 and 2005–2026). Standard Errors are displayed.

BLUNTNOSE GUITARSHARK

Bluntnose guitarshark *Acroteriobatus blochii* is the most common guitarfish along the West Coast, found from the Orange River in Namibia and extending southwards to Cape Point in South Africa (Ebert et al. 2021). They are a relatively small species (up to 96 cm total length) and live in shallow sandy bottom waters (Pollom et al. 2020). They are frequently confused with the lesser guitarshark *A. annulatus* (van Staden et al. 2022). Bluntnose guitarshark are a popular angling fish along the South African coastline where they are found, and within Saldanha Bay, and therefore likely to be caught in the beach seine fishery and potentially misreported which has led to confusion with identification over the years (da Silva et al. 2015).

Our data show some interannual variation in the abundance of bluntnose guitarshark caught in Saldanha Bay with periodic high abundance, but with most years having an average of between 0.1 and 1 guitarfish per haul (Figure 9.12). In 1994 a total of 184 individuals were caught, the highest catch of any year, with 79% of the total catch found in the Lagoon, including one haul of 91 individuals caught in Kraalbaai. The 2007 and 2019 surveys had similarly high abundance (84 and 64 individuals caught, respectively), with the largest catches in Bluewater Bay, Small Bay. Since the most recent peak in 2019, guitarshark abundance has been on the lower end of the average range, mostly below 0.2 fish/haul (except 2021 which was 0.4 fish/haul) (Figure 9.12). Last year (2025) marked the first year in the history of the SOB seine net surveys in which zero individuals were caught. In 2026 a single individual was caught, in the Lagoon (Figure 9.12).

Seine net surveys undertaken by University of Cape Town students between March and early May between 2017–2019 and 2024–2026, while less standardised than the SOB surveys, have recorded similar patterns of reduced bluntnose guitarshark abundance in recent years, and also caught zero individuals in 2025, and 2026 (CG Attwood *pers. comm.* 2026)

However, the latter results do not indicate an emergency, as catches in most years have been relatively low. Furthermore, the Western Cape Shore Angling Association Interclub Competition held in St Helena Bay (~50 km to the north of Saldanha Bay) on 16 November 2024 recorded catches of well over 1 000 bluntnose guitarshark, mostly females (A de Villiers *pers. comm.* 2025). This difference could be due to the timing of these events, or that conditions (e.g., water temperature) in Saldanha Bay may currently be less suitable for the species, that water conditions during the SOB sampling period were not suitable the last few years, or alternatively, that pressures in St Helena Bay are less severe. Continued monitoring will be essential in the coming years, alongside further investigation into potential drivers of the apparent reduced abundance of bluntnose guitarshark in the Saldanha-Langebaan system.

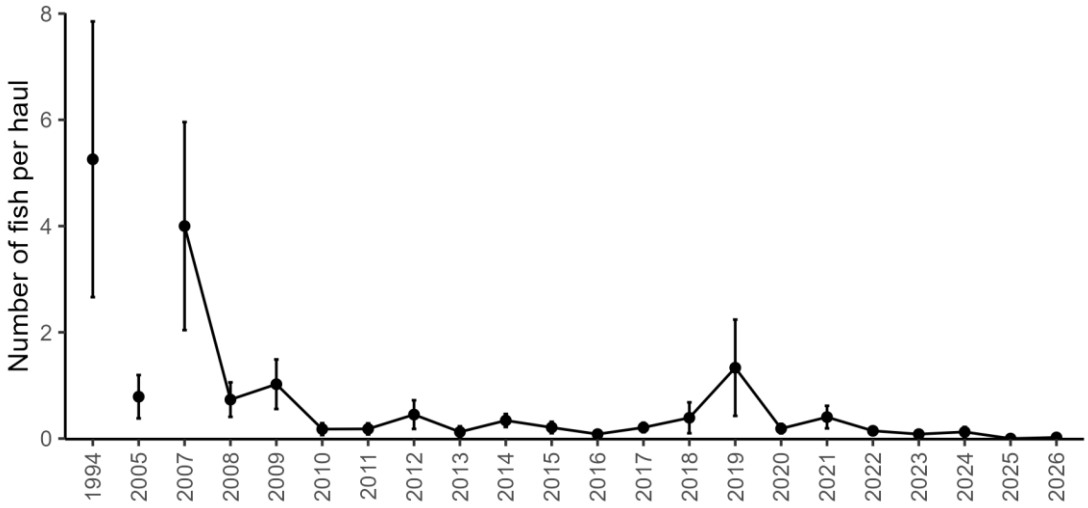


Figure 9.12. Average annual catch per haul of bluntnose guitarshark caught in beach seine net sampling in Saldanha Bay and Langebaan Lagoon (1994 and 2005–2026). Standard Errors are displayed.

9.3.4 SUPPORTING FISH POPULATION ASSESSMENTS

BAITED REMOTE UNDERWATER VIDEO (BRUVS)

Since February 2019, SANParks has employed Baited Remote Underwater Video (BRUV) surveys to monitor fish, sharks and rays within the Langebaan Lagoon MPA. BRUVs are a non-destructive and cost-effective monitoring tool that is widely utilised worldwide. This monitoring programme assesses the diversity and relative abundance of fish, sharks and rays, providing crucial data that inform management decision-making across the different zones of the Langebaan Lagoon MPA. To date, 231 hours of BRUV video have been recorded across all three MPA zones (Zones A, B and C, Figure 10.2). Surveys were conducted during both summer and winter from 2019 to 2022. However, from 2023 onwards, only summer surveys were undertaken due to poor visibility during winter. BRUVs were also deployed outside the MPA during the summer and winter of 2019, but the very poor visibility prevented continued sampling outside of the MPA. A total of 27 species of fish, sharks and rays from 20 families have been detected (Table 10.3), with steentjie, white stumpnose and Cape silverside being the most frequently observed. Generally, summer months had greater abundance and diversity than winter for most species and juvenile fish. This was likely attributed to higher average temperatures in

summer (22°C) than during winter (15°C) months. This trend was particularly evident for steentjies and white stumpnose. Mono-BRUV systems were used from 2019 to 2022. In 2023, through a collaboration with the South African Environmental Observation Network (SAEON), SANParks upgraded the mono-BRUVs to stereo-BRUV systems. This allowed for the remote measurement of fish length, an additional metric to assess the status of fish stocks. A longer time series will allow investigation of the influence of the level of protection on the abundance and diversity of fish, shark and ray species within the MPA (SANParks pers. comm. 2026).

E-DNA

eDNA sampling was undertaken by Anchor at 10 select sites during 2023, 2025 and 2026. eDNA is DNA that is collected from a variety of environmental samples (e.g., water or sediment) rather than directly sampled from an individual organism. As organisms interact with the environment, DNA is expelled and accumulates in their surroundings. Examples of eDNA sources include, among other things, faeces, mucus, gametes, shed skin, carcasses and hair. Sampling using eDNA has the advantage of being able to detect organisms in an area that are otherwise not found with other sampling methods, because they are, for example, evasive, elusive, or endangered. This is particularly useful in an aquatic environment, where once-off "snapshot sampling" using traditional methods (e.g., grabs, cores, nets, or lines) may not necessarily reveal all species present.

Filtered water samples collected from each site were submitted to an established eDNA service provider, Nature Metrics in the United Kingdom, for analysis of vertebrate genetic sequences using markers from their "vertebrate" and "fish" pipelines[6]. Species level resolution using eDNA/metabarcoding is, however, dependent on the available databases of DNA sequences (which is growing continuously) and the sequence matching algorithms employed. The identifications range in terms of the proportion of DNA sequences detected within each sample. This signal may be linked to abundance of species in the environment but only as a coarse measure because the signal is also impacted by biological (e.g., biomass, life stage, activity, body condition), environmental (e.g., temperature, pH, salinity, conductivity), and technical factors.

The results of the eDNA sampling indicated the presence of 24 taxa identified to species level and a further 23 taxa of fish, chimaeras, and elasmobranchs identified to a higher taxonomic level, of which at least 20 were also recorded during beach seine net sampling for juvenile fish (Table 9.3). The eDNA sampling also picked up many pelagic and reef taxa that are not targeted by the seine net sampling, and provide insight into other species that may have been present in the Bay. Interpretation is undertaken with caution, however, as the detection of two *Merluccius* species (hake) and *Lophius* sp. (monkfish), which are offshore demersal species, is almost certainly due to effluent discharge from the Sea Harvest fish processing plant. Additionally, the presence of a tilapia *Oreochromis* sp. is likely due to outflow from rivers, as they are freshwater species.

[6] A metabarcoding pipeline converts millions of raw DNA sequences from high-throughput sequencing into ecological data by filtering, trimming, merging reads, removing errors, clustering similar sequences into molecular Operational Taxonomic Units (OTUs), and matching them to reference databases. The result is a table showing how many sequences from each sample belong to each OTU.

Table 9.3. Fish and elasmobranch taxa recorded in eDNA and/or BRUV surveys from Saldanha Bay and Langebaan Lagoon. Taxa in bold were also recorded in State of the Bay seine net samples. IUCN Red List classification is provided (EN = Endangered, VU = Vulnerable, NT = Near Threatened, DD = Data Deficient, LC = Least Concern, NE = Not Evaluated) (IUCN 2025).

Group & Family	Taxon	Common Name	IUCN Red List status	eDNA	BRUV
Elasmobranchs					
Carcharhinidae	*Carcharhinus brachyurus*	Copper shark	VU		x
Hexanchidae	*Notorynchus cepedianus*	Broadnose sevengill shark	VU	x	x
Pentanchidae	*Haploblepharus pictus*	Dark shyshark	LC		x
	***Haploblepharus* sp.**	shyshark sp.	-		x
Rajidae	*Rostroraja alba*	White skate	EN		x
	Raja straeleni	Biscuit skate	NT		x
Rhinobatidae	***Acroteriobatus* sp.**	Guitarshark sp.	-		x
Scyliorhinidae	*Poroderma africanum*	Pyjama shark/ striped catshark	LC		x
Triakidae	*Mustelus mustelus*	Common smooth hound shark	EN		x
Holocephali					
Callorhinchidae	*Callorhinchus capensis*	Cape elephantfish/ St Joseph shark	LC	x	
Teleosts					
Ariidae	*Galeichthys ater*/ ***Galeichthys feliceps***	Black seacatfish/ White barbel	NE	x	x
Atherinidae	***Atherina breviceps***	Cape silverside	NE	x	x
	***Atherinidae* sp.**	-	-	x	
Batrachoididae	*Batrichthys apiatus*	Snakehead toadfish	LC		x
Blenniidae	***Parablennius* sp.**	Combtooth blennies	-	x	
Carangidae	*Decapterus macarellus*/ *Decapterus muroadsi*	Mackerel scad/ Amber-striped scad	LC	x	
	Lichia amia	Garrick/leervis	LC		x
	Seriola lalandi	Yellowtail kingfish	LC	x	
	Seriola lalandi/ *Seriola quinqueradiata*	Yellowtail kingfish/ Japanese amberjack	LC	x	
	Trachurus trachurus	Atlantic horse mackerel	VU	x	x
	Trachurus capensis	Cape horse mackerel	LC		x
Chaetodontidae	*Chaetodon marleyi*	Doublesash butterflyfish	LC		x
	Chaetodon sp.	Butterflyfish sp.		x	
Cheilodactylidae	*Cheilodactylus* sp.	Fingerfish sp.	-	x	
	Cheilodactylus fasciatus	Redfingers	LC	x	
Cichlidae	*Oreochromis* sp.	Tilapia sp.		x	
Clinidae	***Clinus* spp.**	Klipfish spp.	-		x
Clupeidae	***Sardinops sagax***	Pilchard/sardine	LC	x	
Cynoglossidae	***Cynoglossus* sp.**	Tonguefish sp.		x	
Dussumieriidae	***Etrumeus whiteheadi***	Redeye roundherring	LC	x	
Ehiravidae	*Gilchristella aestuaria*	Estuarine round herring	LC		x
Engraulidae	*Engraulis encrasicolus*	Cape anchovy	LC	x	
	Engraulis encrasicolus/ ***Engraulis japonicus***	Cape anchovy/ Japanese anchovy	LC	x	

Group & Family	Taxon	Common Name	IUCN Red List status	eDNA	BRUV
Gempylidae	***Thyrsites atun***	Snoek	NE	x	
Gobiesocidae	*Apletodon* sp. (likely *Apletodon pellegrini*)	Cubby clingfish	LC	x	
	Diplecogaster sp. (likely *Diplecogaster megalops*)	Bigeye clingfish	LC	x	
Gobiidae	***Gobiidae* spp.**	Gobies	-	x	
	Psammogobius knysnaensis	Knysna sandgoby	LC		x
Haemulidae	*Pomadasys olivaceus*	Olive grunt	LC		x
Lophiidae	*Lophius* sp.	Monkfish sp.		x	
Merlucciidae	*Merluccius capensis*	Shallow-water Cape hake	LC	x	
	Merluccius paradoxus	Deep-water Cape hake	NE	x	
Mugilidae	*Chelon dumerili*	Grooved mullet	DD	x	
	Chelon richardsonii	Southern mullet/harder	NE	x	x
	Chelon richardsonii/ Chelon tricuspidens	Southern mullet/ Striped mullet	-	x	
	Mugil cephalus	Flathead grey mullet	LC	x	
	Osteomugil cunnesius	Longarm mullet	NE	x	
Ophichthidae	*Ophisurus* sp. (likely *Ophisurus serpens*)	Sand snake eel	LC	x	
Ophidiidae	Ophidiidae sp.	Cusk eel sp.		x	
Pomatomidae	***Pomatomus saltatrix***	Bluefish/Elf/Shad	VU	x	x
Sciaenidae	*Argyrosomus inodorus*	Silver kob	VU	x	x
	Argyrosomus sp.	-	-	x	
Scombridae	*Scomber* sp.	Mackerel sp.	-	x	
Serranidae	*Serranus cabrilla*	Comber	LC		x
Soleidae	***Heteromycteris capensis***	Cape sole	NT		x
	***Soleidae* sp.**	Sole sp.		x	
Sparidae	*Diplodus puntazzo/ Diplodus sargus*	Sheephead bream/ White seabream	-	x	
	Diplodus sargus	White seabream	LC	x	
	Diplodus capensis	Blacktail	LC		x
	***Diplodus* sp.**	-	-	x	
	Lithognathus aureti	West coast seabream	EN		x
	Lithognathus lithognathus	White steenbras	EN		x
	Lithognathus mormyrus	Sand steenbras	LC		x
	Pachymetopon blochii	Hottentot seabream	LC		x
	Rhabdosargus globiceps	White stumpnose	VU	x	x
	Sarpa salpa	Strepie	LC		x
	Sparidae sp.	-	-	x	
	Sparus sp.	-	-	x	
	Spicara axillare	Windhaan	LC		x
	Spondyliosoma cantharus	Black seabream	LC	x	
	Spondyliosoma emarginatum	Steentjie	LC		x

Group & Family	Taxon	Common Name	IUCN Red List status	eDNA	BRUV
	***Spondyliosoma* sp.**	-	-	x	
	Stenotomus sp. *none found in SA	Porgies/scups	-	x	
Syngnathidae	***Syngnathus* sp.**	pipefish	-	x	x
Tetraodontidae	*Arothron* sp. (likely *Arothron inconditus*)	Bellystriped blaasop		x	
	Torquigener sp.	Blaasop sp.		x	
Triglidae	*Chelidonichthys kumu/ Chelidonichthys lucerna*	Bluefin gurnard/ Tub gurnard	-	x	x
	***Chelidonichthys* sp.**	-	-	x	

9.3.5 ECONOMICALLY IMPORTANT INVERTEBRATES

SQUID

Anchor divers sighted Cape hope squid *Loligo reynaudii* eggs attached to the ADZ nitrate sensor in Small Bay on three occasions in 2026 (Figure 9.13). This species of squid is also known as 'chokka' and is distributed around the southern African continental shelf from Namibia to Port Alfred in the Eastern Cape, with an additional population in southern Angola and are more abundant on the South and East coasts than the West Coast (Augustyn 1990). Chokka squid is targeted by the jig fishery operating between the Cape of Good Hope and Port Alfred, employing approximately 3 000 people. The squid are almost exclusively exported to Europe and the fishery is South Africa's third largest fishery in terms of monetary value (DFFE 2025).

Figure 9.13. Left: *Loligo reynaudii*. Photo: © josepvilanova. Right: Squid eggs found attached to the nitrate sensor in Small Bay in March 2026. Photo: Robyn Payne.

On the West Coast, the species is distributed discontinuously north of Cape Columbine occurs at greater depths (to ~350 m) than on the South Coast. Adult demersal distribution on the West Coast is strongly influenced by bottom temperature, dissolved oxygen and current regime, conditions that are more extreme there than on the south coast (Roberts 2005).

Surveys of egg-capsule distribution between Port Nolloth and Port Alfred found that reproduction was confined to the eastern and western Agulhas Bank, particularly the inshore-to-mid-shelf region between Plettenberg Bay and Port Alfred, extending to depths of ~270 m near the shelf edge ((Augustyn 1990, Sauer et al. 1992, Sauer & Smale 1993, Roberts et al. 2002). The South and East Coast population has an abundance of reproductively mature adults, while on the West Coast the population consists predominantly of immature and maturing subadults (Augustyn 1991, Roberts 2005). The West Coast therefore functions primarily as a nursery and feeding ground, with juveniles recruiting there after spawning on the Agulhas Bank (Olyott et al. 2007).

Until now, spawning has not been recorded on the West Coast, and only a few records as far west as False Bay have been reported by fishers (Roberts et al. 2012) Given the anomaly of squid egg sightings on the West Coast, eggs from the nitrate sensor (Figure 9.13) were sampled and submitted to a laboratory for genetic analysis, which revealed that the species was indeed *Loligo reynaudii*. So far, eggs have not been sighted on any other surface in the Bay, raising the question as to why the nitrate sensor is subject to repeat spawning events. However, no surveys have been undertaken to determine the extent of egg distribution within the bay.

9.4 CONCLUSIONS AND RECOMMENDATIONS

Overall fish abundance fluctuates substantially between years, mainly because overall abundance is dominated by shoaling species such as harders and Cape silversides, whose abundance can vary enormously. The median fish abundance in Langebaan Lagoon in 2026 was similar to the long-term median for this region, whereas both the Small Bay and Big Bay 2026 medians were roughly double their long-term median. Diversity of seine net catches in 2026 remained lower than average in Small Bay and Langebaan Lagoon, with Small Bay showing a long-term decline. However, diversity in Big Bay increased since 2025 and was only slightly below the long-term average.

Over most of the sampling record, the diversity of seine net samples in Big Bay and Langebaan Lagoon has fluctuated with the stochastic presence of rare species in catches, but with no consistent trend indicating changes in habitat quality. On the other hand, diversity of seine net catches in Small Bay has declined slightly over time and this is partly attributable to the loss of elasmobranchs like smooth hound sharks, eagle rays *Myliobatis aquila* and bluntnose guitarsharks, suggesting that the Small Bay habitat quality may have declined. However, these same elasmobranch species have also become less common in Big Bay and the declines in overall abundance of these species in the system are probably related to the broader stock status throughout these species ranges. Smooth hound sharks (EN) and eagle rays (CR) are classified on the IUCN Red List (IUCN 2025). Other Red Listed elasmobranchs present in the system that have been recorded in earlier seine net surveys and/or BRUV footage and/or eDNA sampling include: puffadder shyshark *Haploblepharus edwardsii*, (EN), Blue stringray *Dasyatis chrysonota* (NT), Thornback skate *Raja clavata* (NT), broadnose sevengill shark *Notorynchus cepedianus* (VU), white skate *Rostroraja alba* (EN), and biscuit skate *Raja straeleni* (NT). Elasmobranchs typically have life history characteristics such as slow growth, long lifespans delayed maturity and low fecundity that make them particularly vulnerable to anthropogenic pressures–especially fishing even when only landed as by catch. The presence of all these Red Listed elasmobranch species in the Saldanha-Langebaan system highlights the importance of the unique habitat in the region and necessity of good

management to protect them. Steps are currently underway to propose the Saldanha Langebaan system as an Important Shark and Ray Area (A. Kock, SANParks, Personal Communication).

The 2026 survey results were consistent with most of the seine net survey history, where fish abundance at sites within Small Bay and Langebaan Lagoon MPA is typically higher than that recorded at the more exposed Big Bay sites. This reflects the nursery value of these more sheltered, regions of the system that have characteristics (warmer, calmer and often enriched) that enhance juvenile fish survival and growth. The relationship between surfzone habitat exposure and juvenile fish communities is well established (Clark 1997). It is pertinent to note that most of the Lagoon habitat is protected within the West Coast MPA, whilst the Small Bay habitat is not, and it is under increasing pressure from anthropogenic impacts. The location of Small Bay at the northern end of the system, and the prevailing southerly winds during the spring-summer spawning season also means that a lot of the planktonic fish larvae will by default need to utilize the Small Bay nearshore nursery habitats. Ongoing efforts to maintain these Small Bay nursery habitats in a good state of ecosystem health is essential for the Bay's fish stocks.

Maintenance of habitat quality is critical, but the greatest present threat, and driver of change in the Saldanha-Langebaan fish community to date, is consumptive utilization i.e. fishing, that has reduced the adult population of targeted species throughout the system. The protection from exploitation and reduced disturbance at some sites due to the presence of the Langebaan Lagoon MPA is therefore likely to be having a positive effect on the maintenance of juvenile fish populations. 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 further access to Zone B for commercial gillnet 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 not to have 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 et al. 2019).

The significant declines in juvenile white stumpnose abundance that occurred throughout the system over the period 2007–2026 suggested that the protection afforded by the Langebaan Lagoon MPA was not enough to sustain the fishery in the face of prevailing high effort levels. Evidence from seine net surveys certainly suggest that recruitment overfishing has occurred. The annual seine net surveys provided an early warning of poor recruitment and should have allowed for timeous adjustments in fishing regulations to reduce fishing mortality on weak cohorts and preserve sufficient spawner biomass. Unfortunately, despite repeatedly expressing concern about the collapse of white stumpnose recruitment in State of the Bay Reports since at least 2013, and supporting the implementation of the harvest control measures recommended by Arendse (2011); namely a reduction in bag limit from 10 to 5 fish per person per day and an increase in size limit from 25 cm TL to 30 cm TL, the warning calls were not heeded. A statistically comprehensive analysis of fishery dependent and survey data confirmed the collapse of the Saldanha-Langebaan white stumpnose stock and the fishery yield in recent years is a fraction of its historical peak or potential (Parker et al. 2017). A slight

increase in juvenile white stumpnose in the seine net samples has occurred in recent years (2021, 2024–2026). It is likely that the protection of spawner biomass in the Langebaan Lagoon MPA and the high fecundity of female white stumpnose, in conjunction with the reduced targeting of the species has allowed for some recovery of the stock, but it is not clear whether any small recovery will be maintained. To facilitate recovery, it remains prudent to lobby for increased harvest control measures for this important fishery and in 2025 the WQT submitted a motivational letter to the minister of the DFFE calling for improved management of the white stump fishery.

Beach seine net surveys have also revealed some concerning declines in elf recruitment to surf zone nurseries, and it is recommended that this should also be carefully monitored in future. Encouragingly the 2023–2026 surveys also show signs of potential recovery in elf recruitment, but like white stumpnose, it is not clear whether this will be sustained. Interpretation of the recruitment signal of exploited fish species would be greatly enhanced if there was concurrent and ongoing monitoring of recreational catch and effort in the system. Only commercial linefishers are required to submit catch returns and, as most of the white stumpnose and elf fishing effort is recreational, there is a substantial gap with respect to CPUE data for this sector. Such data would provide another direct line of evidence as to the status of exploited fish stocks in the Saldanha-Langebaan system.

The monetary value of the recreational fishery in the Saldanha-Langebaan system should not be regarded as regionally insignificant as a lot of the expenditure associated with recreational angling is taking place within Langebaan Lagoon and Saldanha Bay itself. Furthermore, the once popular white stumpnose fishery was undoubtedly a major draw card to the area and has probably contributed significantly to the residential property market growth the region has experienced. These benefits should be quantified by an economic study of the recreational fisheries. The value of the Bay and Lagoon as a fish nursery and the economic value of the resultant fisheries could then be quantitatively considered when the environmental impacts of the proposed future developments in the region are assessed.

The monitoring record from the annual seine net surveys will prove increasingly valuable in assessing and mitigating the impacts of future developments on the region's ichthyofauna. Extending the seine net monitoring record would also facilitate analysis of the relationship between recruitment to the near shore nursery habitat and future catches in the commercial and recreational fisheries in the Bay. As recreational fishing effort continues to increase, at some point fishing mortality will need to be contained, if the fisheries are to remain sustainable. Many of the key fisheries assessed here are overexploited nationally, and Saldanha Bay is replicating this with evidence of poor recruitment when compared to historical data and in the light of increasing pressures around the Bay. This is particularly important given the importance of this system in providing refuge and a nursery area for many fish species in a region where comparable habitats do not exist.

Regional species-specific fishery management has been implemented elsewhere in South Africa (e.g. Breede River night-fishing ban to protect dusky kob *Argyrosomus japonicus*). 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 DFFE, and we think this approach would work well in the Saldanha-Langebaan system. 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 the bay's fishery. However, without appropriate management, it is likely that when the stronger year classes observed for white stumpnose, elf and harders in recent seine net surveys (2021–2026) recruit to the fishery, fishing effort will again increase in response, and lasting recovery will not be maintained. We also support the recommendation of Horton et al. (2019) for a reduction in harder fishing effort and gear changes (increase in minimum mesh size) to facilitate stock recovery. Short term reductions in fishing mortality will have an economic cost but will yield substantially greater socio-economic benefits for fishers in the medium to longer term as the sustainable catch from an optimally exploited fish stock greatly exceeds that from a collapsed stock.

As reef surveys were not undertaken in 2026, no new data on abalone or rock lobster were available. However, chokka squid eggs were observed for the first time on numerous occasions, attached to the nitrate sensor in Small Bay. This species generally spawns on the South and East Coasts, with only a few records of squid eggs occurring as far west as False Bay, and as such this is a highly anomalous occurrence.

Psammogobius knysnaensis

10 BIRDS AND SEALS

10.1 BIRDS OF THE SALDANHA BAY

10.1.1 NATIONAL IMPORTANCE

The Saldanha Bay 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 (currently listed as Critically Endangered under International Union for Conservation of Nature (IUCN) Red List criteria), Cape Gannet (Endangered), Cape Cormorant (Endangered), White-breasted Cormorant, Crowned Cormorant, Bank Cormorant (Endangered), Kelp and Hartlaub's Gulls, Caspian Tern and Swift Tern (IUCN 2025).

The Department of Forestry, Fisheries and the Environment (DFFE) conducts bird counts on all islands to track population trends of each of these species over time. Each island is normally 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. Normally, all islands are visited roughly three times per calendar year except for Malgas Island (nine times) and Vondeling Island (less than three times due to accessibility) (Rob Crawford, Department of Environmental Affairs *pers. comm.* 2016). Due to the COVID-19 lockdowns, surveys only took place at four of the islands (Jutten, Malgas, Meeuw and Schaapen Islands) and no counts available for Cape Gannet, Caspian Tern, Swift Tern and White-breasted Cormorants. The normal sampling events took place during 2021–2024 (five visits to the five main bird islands) and these data are considered to be valid estimates of the breeding populations on these islands (Makhado et al. 2025). However, no counts have been conducted on Caspian and Marcus Islands since 2020.

Monitoring conducted in terms of the Aquaculture Development Zone (ADZ) EMPr was expanded in October 2022 to include counts of Endangered, Threatened and Protected (ETP) species present within 200 m of aquaculture infrastructure in Saldanha Bay. These counts were initially undertaken opportunistically by mariculture operators, but since January 2024 counts are conducted by a boat-based scientific team that surveys all mariculture infrastructure in Small Bay and Big Bay once every two months throughout the year. Over the period January 2024 to December 2025 these surveys have recorded a total of 26 970 individual ETP animals representing three marine mammal taxa (Heaviside's dolphins, humpback whales and Cape Fur Seals) and 18 bird species. Most ETP species are encountered within Small Bay where birds and seals utilize mussel rafts as haul out and roosting habitat (Figure 10.1). After Cape Cormorants, Kelp Gulls are the most frequently observed seabirds on mussel rafts and this species has also been regularly observed feeding on discarded biofouling during harvesting or maintenance operations, as well as breeding on the rafts. Hartlaub's Gulls, Crowned Cormorants, Common and Terns (Sandwich, Common and Swift) are also regularly encountered roosting on mariculture infrastructure (Figure 10.1). These surveys indicate that the artificial habitat created by mariculture operations is regularly utilised by sea birds and

seals as resting areas in the Saldanha Bay area, but it is not extensively utilised for breeding and is therefore not equivalent to the island breeding habitat.

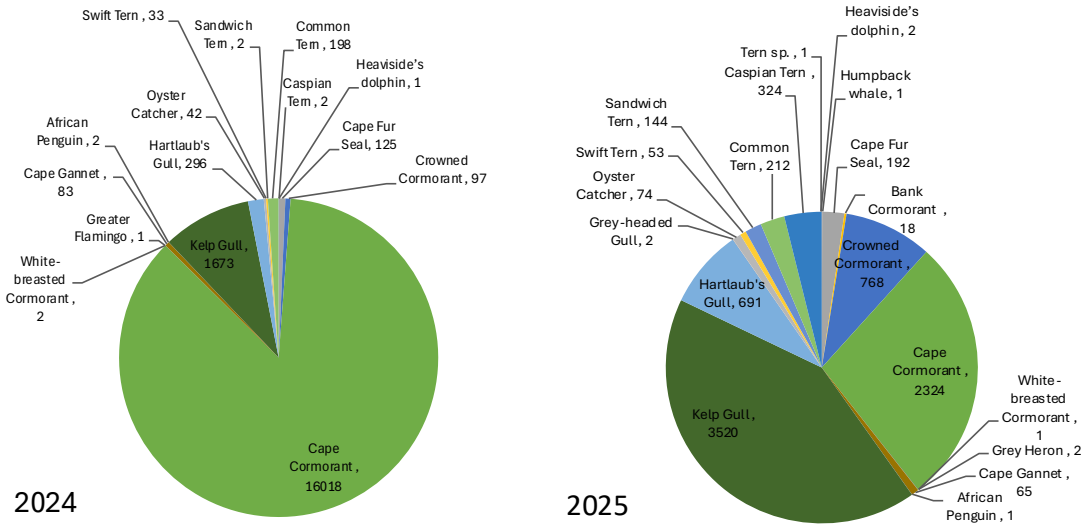
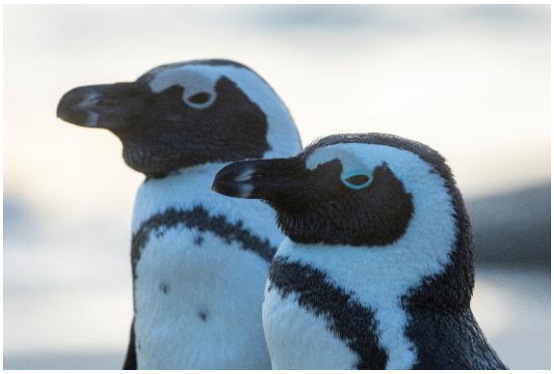


Figure 10.1. Total count of ETP species within 200 m of mariculture infrastructure within Saldanha Bay in 2024 and 2025 (6 counts/year)

10.1.2 ECOLOGY AND STATUS OF THE PRINCIPAL 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. 2005b). The species was up-listed to Critically Endangered in July 2024 under the IUCN Red List due to 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, Crawford 2009, Pichegru et al. 2009, Crawford et al. 2011, 2014, Weller et al. 2014, 2016, de Moor & Butterworth 2015, Grémillet et al. 2016, Parsons et al. 2016, Sherley et al. 2020, BirdLife International 2023). The Namibian population collapsed in tandem with the collapse of its main prey species, the sardine *Sardinops sagax* (Ludynia et al. 2010b). 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. 2005a, Sherley et al. 2020).

The global population of African Penguins (including birds breeding on four Namibian islands) hit a historical low of ~17 700 breeding pairs in 2019 with a high probability of having declined by almost 65% since 1989 (Sherley et al. 2020). The latest counts from 2023 show the decline continuing, with an estimated breeding population of ~8 750 pairs

in South Africa and just ~1 200 in Namibia bringing the global population to <10 000 pairs for the first time (Sherley et al. 2024). Nationally, in South Africa, the African Penguin population declined from an average of 48 000 pairs over the period 1979–2004 a record low of just 9 900 pairs in 2021 (Crawford et al. 2014, Sherley et al. 2020, 2024, Boersma et al. 2020).

West Coast penguin colonies (north of Cape Town) have fared the worst in South Africa, with an unsustainable average annual decline of 10% over the last 20 years (Sherley et al. 2020). The population decline in this region is now equivalent to 75.3% over the preceding 30 years or three generations (Sherley et al. 2024). This is thought to be linked to a distribution shift of their main prey species sardines and anchovies (see below), with the Eastern Cape penguin colonies holding ~41% of the national breeding population in 2020, up from ~27% in 1979 (Sherley et al. 2020). Over the last five years, however, the population in the Eastern Cape has plummeted from around 10 000 pairs over the period 2003–2015 to just ~2 540 pairs in 2023 (Sherley et al. 2024).

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 22 pairs in 2025 (Figure 10.2). DFFE counts during the 2025–26 spring-summer breeding period recorded 17 pairs on Vondeling Island and 3 pairs on Malgas Island and just 2 pairs on Jutten Island pairs (DFFE 2026). The decline in the number of breeding pairs on the Saldanha islands appears to have continued and is consistent with the national downward trend and strongly reinforces the argument that immediate conservation action is required to prevent further losses of these birds.

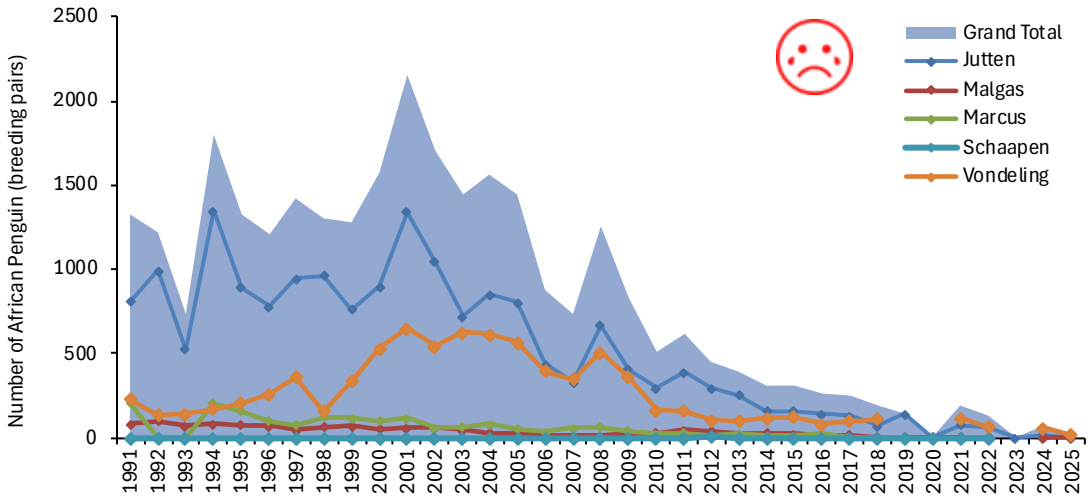


Figure 10.2. Trends in African Penguin populations at Jutten, Malgas, Marcus and Vondeling Islands in Saldanha Bay from 1991–2025 measured in number of breeding pairs Data source: Seakamela at al. (2026).

In light of the ongoing decline in African Penguin numbers nationally, a Biodiversity Management Plan for the African Penguin was gazetted in 2013, with aims: “To halt the decline of the African Penguin population in South Africa within two years of the implementation of the management plan and thereafter achieve a population growth which will result in a downlisting of the species in terms of its status in the IUCN Red List of Threatened Species”. Despite the successful implementation of many of the actions

listed in the plan, these aims were not attained, and African Penguins in South Africa have continued to decline with an IUCN status upgrade to Critically Endangered in 2024. This has led to several revisions of the Biodiversity Management Plan (Government Gazette No. 42775 18 October 2019, Government Gazette No. 47061 22 July 2022). This latest plan attributes population declines mostly to a scarcity of prey and recommends pelagic fishery exclusion zones around colonies, seasonal closures at penguin feeding grounds before and post moult, oil spill risk management and colony specific management such as predator control. Approximately 25% of the South African breeding population now uses mainland colonies, and this highlights the need to manage both marine (e.g. seals) and terrestrial (e.g. caracal, domestic dogs) predators of penguins (Pichegru et al. 2025).

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 (Whittington et al. 2005a). 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 10.3). 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 10.3). Anchovy biomass also increased from the late 1990s, peaked at over seven 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 10.3). 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 Coasts. 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 10.2, Figure 10.3). In the last decade, however, the estimated sardine biomass along the West Coast declined dramatically, with almost none detected in the 2018–2020 acoustic surveys (Figure 10.3). Anchovy biomass too declined from nearly 4 million tonnes in 2013 to the lowest level in the 40-year data record in 2024, with just 53 000 tonnes recorded on the east coast and almost none found along the West Coast (Figure 10.3). Some recovery in anchovy biomass to the east of Cape Agulhas was recorded in the November 2025 survey, but anchovies remained nearly absent from the west coast. However, penguin survival rates do not appear to be correlated with anchovy biomass, but a strong

correlation between adult penguin survival and sardine stock biomass has been reported in most years of particularly low sardine stock biomass (Robinson et al. 2015b, Crawford et al. 2022). Both these studies found little relationship between adult survival and sardine stock biomass in years when stock biomass was average, or above average.

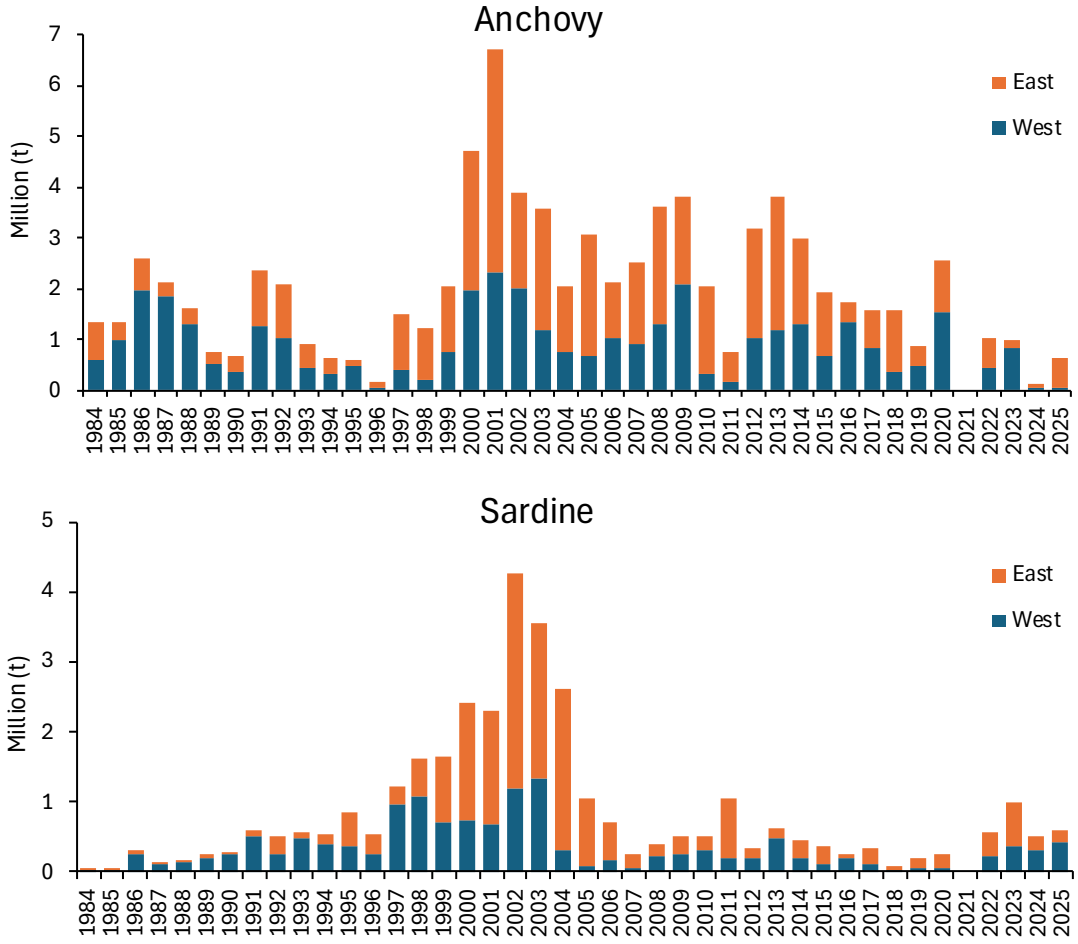


Figure 10.3. Long term trends in the biomass of small pelagic fish (sardine and anchovy) to the west and east of Cape Agulhas based on hydro acoustic surveys conducted annually from 1984–2025 (Data source: Department of Environment, Forestry and Fisheries, 2025).

The estimated biomass of sardines along the West Coast has improved in the last three years and at the time of the November 2025 survey was ~30% of the peak 2003 West Coast biomass (Figure 10.3). Sardine availability therefore appears to have been improving for penguins breeding along the West Coast, including the Saldanha Bay islands, although this may be somewhat offset by the very low anchovy biomass recorded in 2024 and 2025 (Figure 10.3). It is unlikely that this positive trend in sardine biomass will continue given the recent mass mortality of sardines due to the recent outbreak of Pilchard Herpes Virus (PHV) in the South African and Namibian stocks. This virus had devastating impacts on the Australian pilchard stocks (estimated 70% loss) when it first appeared there in the late 1990s and the potential threat to the South African stock has been known for some time (Macey et al. 2016). During July and August 2026 PHV spread rapidly (~40 km per day) throughout the distribution range of South African pilchards with

extensive mortalities reported between Henties Bay in Namibia and Mbotyi on the Wild Coast of South Africa. The full impacts of PHV on the South African pilchard stocks will only be known on completion of the November 2026 acoustic spawner biomass survey undertaken annually by the DFFE. With the Australian PHV outbreaks, seabird diets shifted dramatically and breeding success of both gannets and penguins declined in the subsequent seasons (Bunce et al. 2005, Chiaradia et al. 2010). A similar impact on seabirds should be anticipated with the PHV outbreak here and this will likely exacerbate the already dire situation for African penguins.

There are numerous factors and multiple pressures influencing the negative trend in the African Penguin population size. Some of these pressures occur throughout the species range, while others occur 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, was tested initially 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.

On the 16 September 2022 the DFFE announced closures for small pelagic fishing around the major penguin breeding colonies of Dassen Island, Robben Island, Stony Point, Dyer Island, St Croix Island and Bird Island (www.gov.za/speeches/forestry-fisheries-and-environment). The closures around Dassen Island and Robben Island could benefit penguin colonies on the West Coast as a whole and the Dassen Island closure may benefit penguins nesting on the Saldanha islands as it is within the foraging range of the species (Pichegru et al. 2009). Recently, an expert panel reviewed the results of the Island Closure Experiment and noted that "excluding fishing around island breeding colonies is likely to reduce the rate of decline in the population to a small extent, mediated through improvements in reproductive success" (Punt et al. 2023). Other benefits for penguins, such as increased adult survival was also thought to be a benefit of excluding purse seine fishing around breeding colonies although the Island Closure Experiment did not test for such effects (Punt et al. 2023). Following the completion of the review report, in August 2023 the minister of DFFE announced a decision to extend the purse seine fishing limitations in the waters around important penguin colonies for a minimum of 10 years, with a review after 6 years of implementation and data collection. A recent analysis indicates that the current closed area around Dassen Island is fairly well aligned with the core foraging area of penguins breeding at this colony (whilst it was not well aligned for four other important colonies), although some adjustments of the closed area dimensions would maximise benefits for this endangered species (Mcinnes et al. 2024). Birdlife South Africa and SANCCOB subsequently secured a court order in March 2025 that compelled the DFFE to adjust the closed areas around some of the colonies in line with the recommendations of the Expert Panel. Whether the 2022 purse seine fishing closure around Dassen Island yields benefits for the Saldanha penguins remains to be seen, but it appears to have done nothing to halt the decline in breeding pairs on the Saldanha islands to date.

Several studies have identified additional drivers of African Penguin populations at the colony level. These include oiling and predation by seals and Kelp Gulls, with the importance of fishing and food availability decreasing at small colony size (<3 500

breeding pairs) (Ludynia et al. 2014, Weller et al. 2014, 2016, Butterworth & Ross-Gillespie 2025). It has also been suggested that underwater noise associated with increased vessel traffic may have contributed to a recent decline in penguin numbers at St Croix Island in Algoa Bay and that increased shipping noise may represent an increasing threat to African Penguins in South African waters in general (Pichegru et al. 2022b). This is particularly pertinent to colonies close to large, operational ports such as Saldanha Bay where shipping traffic, and hence underwater noise, is increasing annually (see Chapter 4: Activities and Discharges).

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 decreased dramatically over the period 2010–2018, but the population has increased again since then and is currently at more than 5 000 pairs (see below). Kelp Gull predation on seabird eggs and chicks (mostly gannets and cormorants) on Malgas and Jutten Islands remains common with 1 774 predations recorded in 2023–2024 and 270 in 2024–2025 (Pelican Watch *pers. comm.* 2025). Gull predation on penguin eggs was not recorded by Pelican Watch in the last two seasons but given the very low numbers of penguins on these two monitored islands, this is not unexpected. 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 2013). Similarly, predation by seals (on land and around colonies) had a significant negative impact on these dwindling colonies (Makhado et al. 2006, 2009). Kelp Gull and seal predation certainly remains problematic for any prospects of recovery of penguin breeding on the Saldanha islands. 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).

A large-scale health assessment on the African Penguin found that this species is potentially exposed to conservation-significant pathogens (Parsons et al. 2016). Disease constitutes a major ecological force and has been shown to play an even greater role in threated populations (Parsons et al. 2016). The effect of diseases on seabird population dynamics is currently poorly understood. Both disease outbreaks as well as chronic diseases should both be considered as potential threats to the African Penguin and should be investigated further as part of the conservation efforts (Parsons et al. 2016).

In summary, the initial collapse of the penguin colonies in the area is probably related to food availability around breeding islands and in areas where birds not engaged in breeding are foraging, particularly before and after moulting. However, now that colonies have shrunk so dramatically, the net effect of local conditions at Saldanha Bay are believed to be an increasingly important factor in the continued demise of African Penguin colonies at the islands. The biomass of sardines along the West Coast is showing signs of recovering, but other pressures such as predation and increasing

shipping traffic remain, and the challenges to a population recovery remain great. The realization of any potential benefits to penguin numbers locally and nationally of the small pelagic fishery closures around important breeding colonies will only become apparent over time.

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. Monitoring of seabirds on aquaculture infrastructure has even documented a few Kelp Gulls nesting on the mussel rafts in Small Bay. Kelp Gulls are known to eat the eggs of several other bird species (including African Penguins, Cape Cormorants and Hartlaub's Gulls). Prior to the 1960s, numbers of Kelp Gulls on offshore islands were controlled to protect the guano and egg producing species (Crawford et al. 1982).

Post 1970, Kelp Gull populations were no longer controlled, which, together with the supplementary food provided by fisheries and landfill sites resulted in the doubling of breeding pairs in South Africa by 2002 (Whittington et al. 2016). The introduction and spread of the invasive alien mussel species *M. galloprovincialis* may have also contributed toward the increased availability of food. Kelp Gulls in Saldanha Bay regularly roost on and feed on mariculture infrastructure (Figure 10.1). Consequently, pressures on guano-producing seabird populations shifted from guano exploitation to egg predation by increasing Kelp Gull numbers.

Over the period 2000–2015, the populations on the islands decreased 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 persisted for about 10 years (Figure 10.4). Recent counts show that Kelp Gull numbers are now similar to those at the start of the comprehensive counting with about 6 000 pairs recorded in 2025 (Figure 10.4). 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).

Anthropogenic debris has been found in Kelp Gull nests, especially in colonies located near landfill sites and coastal sites where there was a limited vegetation available for construction (Witteveen et al. 2017). Debris in nests can lead to injury or death as a result of entanglement of chicks and adults. Ropes and straps are used by Kelp Gulls to construct nests, and plastic bags and food wrappers mostly appear to accumulate during the chick rearing period as those items were mostly regurgitated. The fact that the decline of Kelp Gull populations off the Saldanha islands has stopped since 2015 suggests that Pelican predation rather than anthropogenic debris was the main driver of population decline over the period 2000–2015.

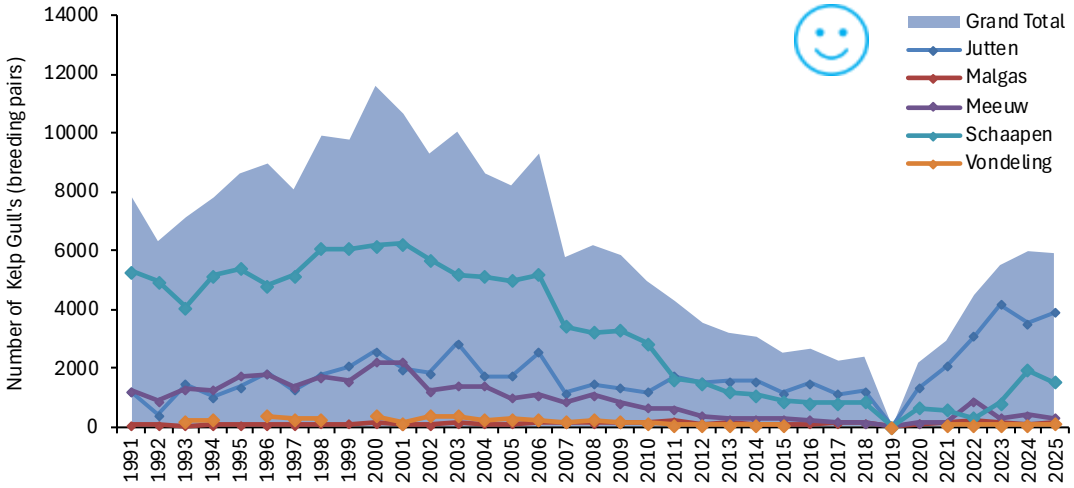


Figure 10.4. Trends in breeding population of Kelp Gulls at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1985–2025 measured in number of breeding pairs Data source: Seakamela at al. (2026).

Hartlaub's Gull *Chroicocephalus 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 & Underhill 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 on Vondeling Island between 1991 and 2006 with just two pairs recorded on Vondeling in 2026. They have also been recorded breeding on Meeuw Island in 1996, from 2002 to 2004, 2012–2014, 2017 and 2020–2025 Hartlaub's Gulls are also consistently recorded on mariculture infrastructure in Saldanha Bay where they roost on rafts and feed on discards during operations (Figure 10.1). 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 & Underhill 2003). Natural predators of this gull are the Kelp Gull, African Sacred Ibis and Cattle Egret, which eat eggs, chicks and occasionally adults (Williams et al. 1990). In Saldanha Bay there is no discernible upward or downward trend over time. The total number of breeding pairs recorded in 2018 was just 36 pairs on Jutten and Malgas Islands, whilst over the period 2019–2025 between 448–996 breeding pairs were recorded with the majority on Jutten, Malgas, Meeuw and Schaapen Islands (Figure 10.5). Overall throughout its range, the population is increasing and the species is not currently facing any threats that are thought to have a significant effect on the population (BirdLife International 2023).

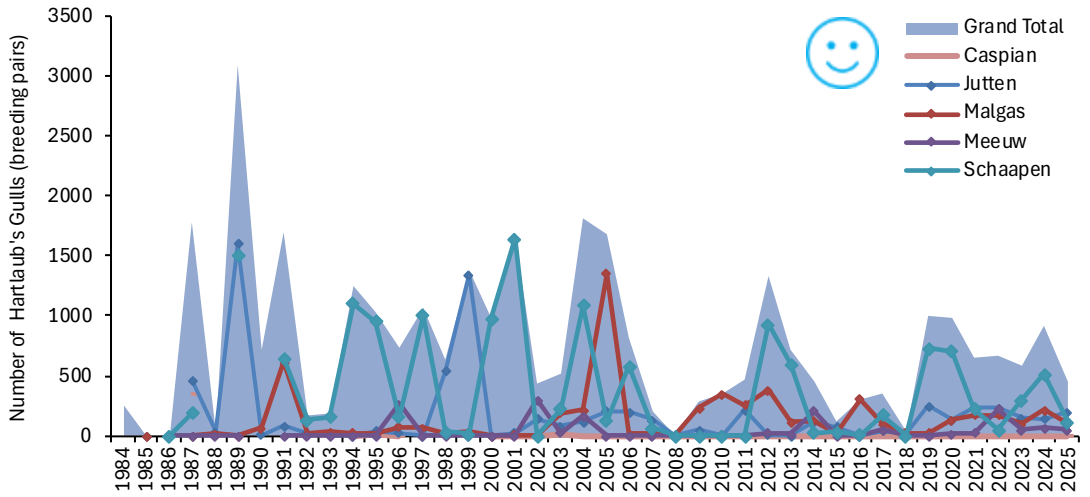


Figure 10.5. Trends in breeding population of Hartlaub's Gulls at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1984–2025 measured in number of breeding pairs. Data source: Seakamela at al. (2026).

The **Greater Crested Tern** *Thalasseus bergii* is a widespread species that occurs as a common resident in southern Africa. The species 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 (Gochfeld et al. 2018). During the breeding season, fish form 86% of all prey items taken, particularly pelagic shoaling fish, of which the Cape anchovy *E. encrasicolus* is the most important prey species. The steady increase in Greater Crested Tern numbers between 2002 and 2005 coincided with a greater abundance of two of their main prey species, sardines and anchovies (Figure 10.6). 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, Greater Crested 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, Crawford et al. 2014).

In Saldanha Bay, Jutten Island has been the most important island for breeding Greater Crested 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 Greater Crested Terns were reported breeding on islands in the Bay for the three years following this, the longest absence on record. Subsequent to this, Greater Crested Terns have bred on the Saldanha islands nearly every year, albeit with the typical, erratic variability. In the last six years, numbers recorded have fluctuated from amongst the highest on record in 2018, to zero in 2020 and 2021, up to around 400 pairs counted on Malgas Island in 2024 and 2025 (Figure 10.6). The low sardine and more recently anchovy biomass on the West

Coast is the most likely cause for the relatively low numbers of breeding Greater Crested Terns on the Saldanha islands during the last five seasons (Figure 10.3, Figure 10.6).

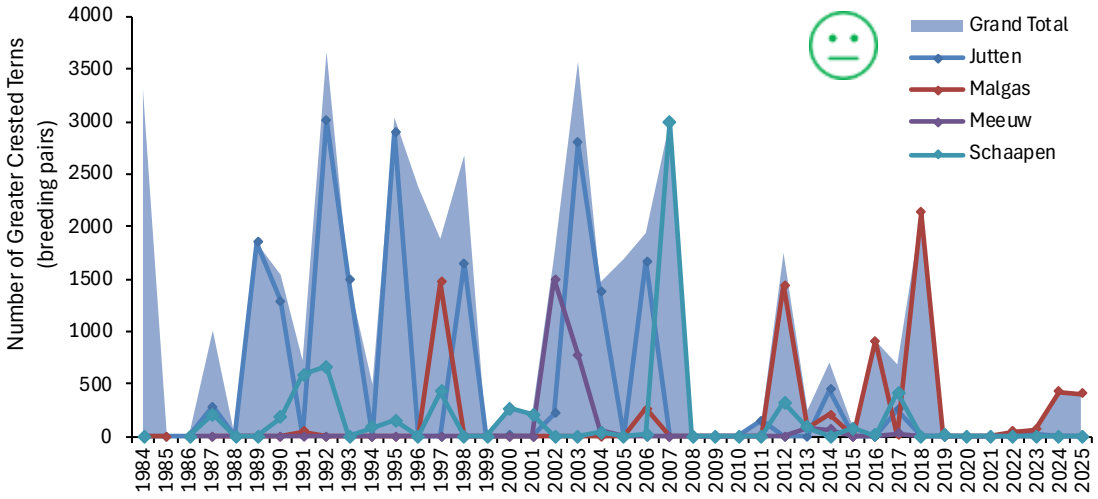


Figure 10.6. Trends in breeding population of Greater Crested Terns at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1984–2024 measured in number of breeding pairs. Data source: Seakamela at al. (2026).

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 Gqeberha. Breeding on islands afford them protection from terrestrial 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 1990s and an erratic decline over the period 1996–2015, which appears to have stabilised since (Figure 10.7). The decline in numbers at Malgas Island coincided with a population increase at Algoa Bay's Bird Island over the same period. The total South African Cape Gannet population appears to respond to the population dynamics of small pelagic fish (particularly sardines), with the number of breeding pairs averaging at 123 000 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, which has resulted in West Coast gannets having to increase their foraging efforts. During the breeding season, they forage in areas with very low abundance of their preferred prey, and feed primarily on low-energy hake trawl fishery discards which constituted 93% of total prey intake (Crawford et al. 2007, 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, Grémillet 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). Competition with purse-seine fisheries in South Africa is associated with negative impacts on foraging success, adult body condition and chick growth rates (Okes et al. 2009, Cohen et al. 2014, Grémillet et al. 2016) but there is little evidence for impacts on the local population size or a decrease in adult survival rate (Crawford et al. 2014). 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 10.7). Nonetheless, a purse seine fishing exclusion zone around important penguin colonies, including Dassen Island, was implemented in 2022 for a 10-year period. Recent increases in the number of Cape Gannets breeding on Malgas Island suggests that the fishery closure is benefitting the species.

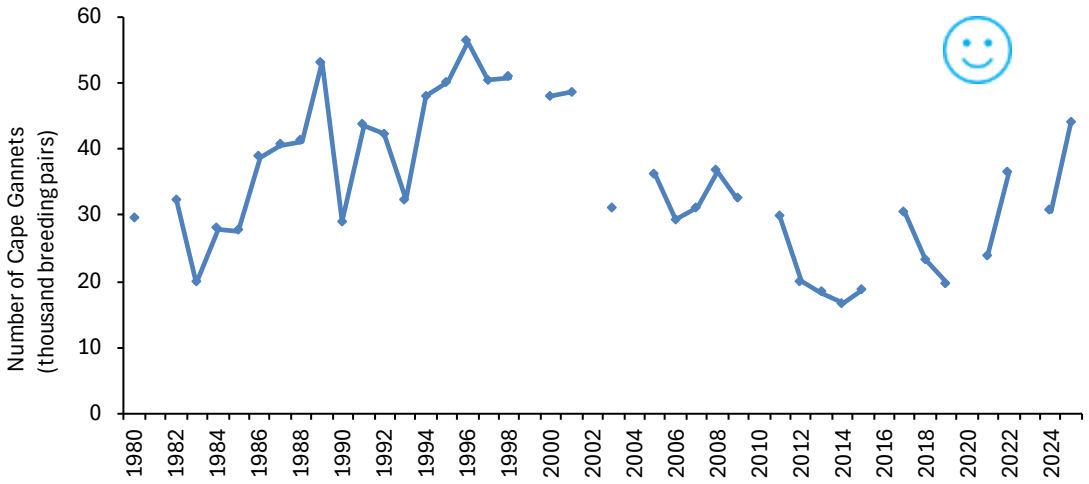


Figure 10.7. Trends in breeding population of Cape Gannets at Malgas Island, Saldanha Bay from 1980–2025 measured in number of breeding pairs. Data source: Seakamela at al. (2026).

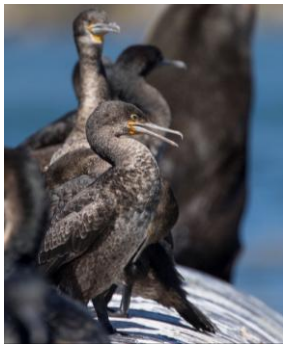
Gannets with their extensive foraging range and diverse diets have proved adaptable to the changes in pelagic fish distribution and nationally, until recently, numbers were thought to have not declined (Crawford et al. 2014). The most recent IUCN Red List Assessment, however, upgraded Cape Gannets from Vulnerable (2017) to Endangered (2018) due to a large population reduction (51.5% between 1956 and 2015) over the past three generations with projected further declines over the next three generations (BirdLife International 2018a). A revised assessment estimated the population decline at 44.7% over the preceding three generations with the balance of evidence suggesting that a global classification of Vulnerable would be the most appropriate status (Sherley et al. 2019). The IUCN Red List status, however, remains Endangered (BirdLife International 2018a). The impacts of global warming including sea level rise and increased frequency and magnitude of storms constitutes a further risk to Cape Gannet colonies due to

coastal flooding that may exacerbated by historical guano mining (BirdLife International 2018a). Direct predation by Cape fur seals, Kelp Gulls and the Great White Pelican constitute further threats to Cape Gannet populations (Makhado et al. 2006, Pichegru et al. 2007, Sherley et al. 2019).

The latter is possibly of greater significance and of more concern at a local level for the Malgas Island Cape Gannet colony, as high rates of predation by Cape fur seals and Kelp Gulls historically accounted for between 1000–2000 gannet breeding failures per season in average years (Pelican Watch *pers. comm*. 2017). Over the last two monitoring seasons (2023–24 and 2024–25) by Pelican Watch teams, 82 seal predations of gannets were observed, whilst gull predation of eggs and chicks (gannets and other seabirds) ranged between 277 and 1 774, (many of these gull predations were on Cape Cormorants that bred on Malgas Island in 2024). 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, Strydom et al. 2022). Seal numbers nationally increased at an average of 3.5% per annum since 1971 until 1993 when aerial census of seal colonies was undertaken (see more below) (David et al. 2003). In Saldanha waters, seal numbers have increased dramatically since 2000 when they started re-colonising Vondeling Island up until 2013 after which numbers have fluctuated around 20 000 (see Section 10.3). 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 fledglings on their first forays from the islands (Pelican Watch *pers. comm*. 2017).

Cape Gannet mortality caused by Cape fur seals amounted to 29–83% of the overall production of fledglings at Malgas Island in the 2000–2006 breeding seasons, despite an ongoing "problem" seal culling programme 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 Gannet population between 2001 and 2006 (Makhado et al. 2006). Culling of problem seals has continued in an effort to improve breeding success (Makhado et al. 2009, Sherley et al. 2019). The effects of this, along with other measures (Dassen Island closure to small pelagic fishing) may be manifest in the observed recovery in Gannet numbers on Malgas Island. Unfortunately, no counts of Cape Gannet breeding pairs were conducted during 2020 and 2023, but the average count over the period 2022, 2024 and 2025 was around 37 100 breeding pairs on Malgas Island, which is 77% of the average number of pairs recorded over the period 1986–2001 (Figure 10.7). There is no consistent trajectory of change in the number of Gannets breeding on Malgas Island over the last two decades and the status of this species in the area appears secure.

Cape Cormorants *Phalacrocorax capensis* are endemic to southern Africa, where they were historically 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, pilchard and Cape horse mackerel *Trachurus trachurus* (Crawford et al. 2016, Masiko et al. 2021).



Key colonies of the Cape Cormorant in South Africa and Namibia have undergone very rapid population declines over the past three generations and the Cape Cormorant has

therefore been up-listed to Endangered (BirdLife International 2018b). Declines are primarily believed to have been driven by collapsing pelagic fish stocks (BirdLife International 2015). However, pelagic fish stocks increased greatly in the late 1990s and early 2000s, and although sardine biomass subsequently crashed, anchovy biomass remained relatively high until 2020, and sardine biomass is showing signs of recovery in the last three years (Figure 10.3). This suggests that other factors are also involved in declining Cape Cormorant numbers. Overall the numbers of Cape Cormorant breeding in South Africa decreased by nearly 57% from ~107 000 pairs in 1977–1981 to 57 000 pairs in 2010–2014 (Crawford et al. 2016). This decline largely occurred along the West Coast north of Dassen Island, whilst the number of breeding pairs at colonies to the south and east of Dassen Island stayed relatively stable or increased in the 2000s (Crawford et al. 2016).

The decrease during the early 1990s is attributed to a combination of factors, including 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 2007, Crawford et al. 2016). Semi-systematic annual counts by the Pelican Watch on Jutten and Malgas Islands, suggests that about between 100–3 000 young Cape Cormorants are predated by seals during the fledging period each year. 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 (Masiko et al. 2021). This makes it difficult to accurately determine population trends on the Saldanha islands. In addition, during outbreaks of avian cholera, tens of thousands of birds die (Crawford et al. 2016). 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 (Crawford et al. 2016).

The Saldanha Bay population has been quite variable since the start of monitoring in 1988, with the bulk of the population residing on Malgas and Jutten Islands in recent years (Figure 10.8). Overall, the number of breeding pairs declined gradually since the 1990s, and although interannual variability is high there appears to be no trend in the last decade. In 2023, a total of only 771 breeding pairs were recorded, representing the lowest level recorded to date (Figure 10.8). The number of breeding pairs on the Saldanha islands recovered well in 2024 and 2025, with around 6 000 pairs counted mostly on Jutten and Malgas Island (Figure 10.8).

White-fronted plover

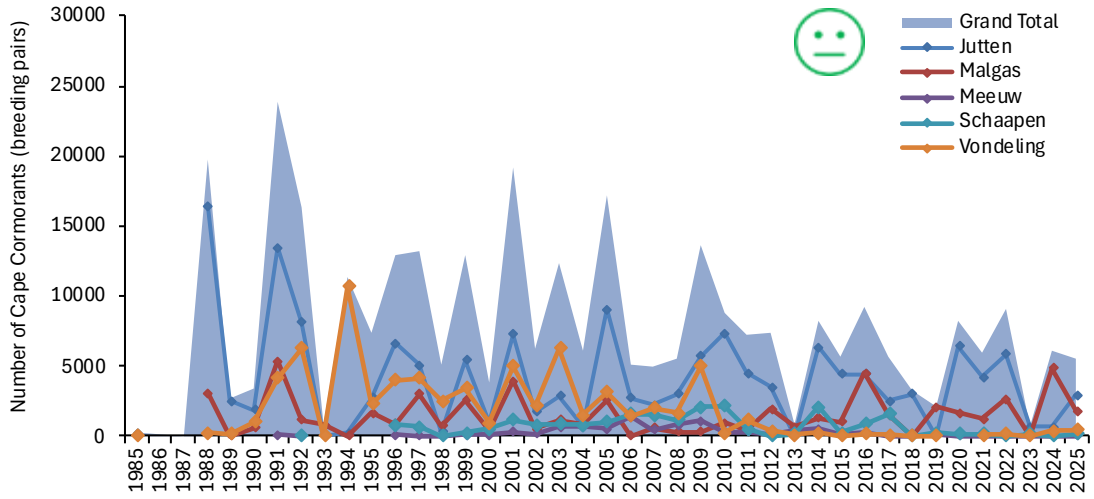


Figure 10.8. Trends in breeding population of Cape Cormorants at Jutten, Malgas, Meeuw Schaapen, and Vondeling Islands in Saldanha Bay from 1980–2025 measured in number of breeding pairs. Data source: Seakamela at al. (2026).

Bank Cormorants *Phalacrocorax neglectus* are endemic to the Benguela upwelling region of southern Africa, breeding from Hollamsbird Island Namibia to Quoin Rock, South Africa. They seldom range farther than 10 km offshore. Their distribution roughly matches that of kelp *Ecklonia maxima* beds. They prey on various species of fish such as the pelagic goby, crustaceans and cephalopods, feeding mainly amongst kelp where they catch West Coast rock lobster, *Jasus lalandii* (Ludynia et al. 2010a, Dyer et al. 2019). The total population decreased from about 9 000 breeding pairs in 1975 to less than 5 000 pairs in 1991–1997 to 2 500 pairs in 2015 (Kemper et al. 2007, BirdLife International 2018c). 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 Northern 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 of mature male rock lobster status now estimated at just 1.3% of pristine biomass and the spawner biomass at just 5.3% of pristine (Cockcroft et al. 2008, DFFE 2025). The degree of depletion of the rock lobster resource is evident in the fact that the allocated Total Allowable Catch to the commercial fishing sectors has not being landed for the last eight years (DFFE 2025). Analysis of bank cormorant diet based on pellets collected at breeding colonies during 1975–1985 and again between 1995–2002 revealed significant declines in the proportion and mean size of rock lobster in the diet (Dyer et al. 2019). Ongoing population declines led to the Bank Cormorant's status being changed from Vulnerable to Endangered (BirdLife International 2018c)

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 has declined steadily since 2003 on all the Saldanha islands where the species used to breed (Malgas, Jutten, Marcus and Vondeling) and only a single pair was recorded breeding on Malgas in 2025. A few Bank Cormorants (maximum count of 11) have been observed roosting on mussel rafts in Small Bay on two occasions during ADZ monitoring (Figure 10.1).

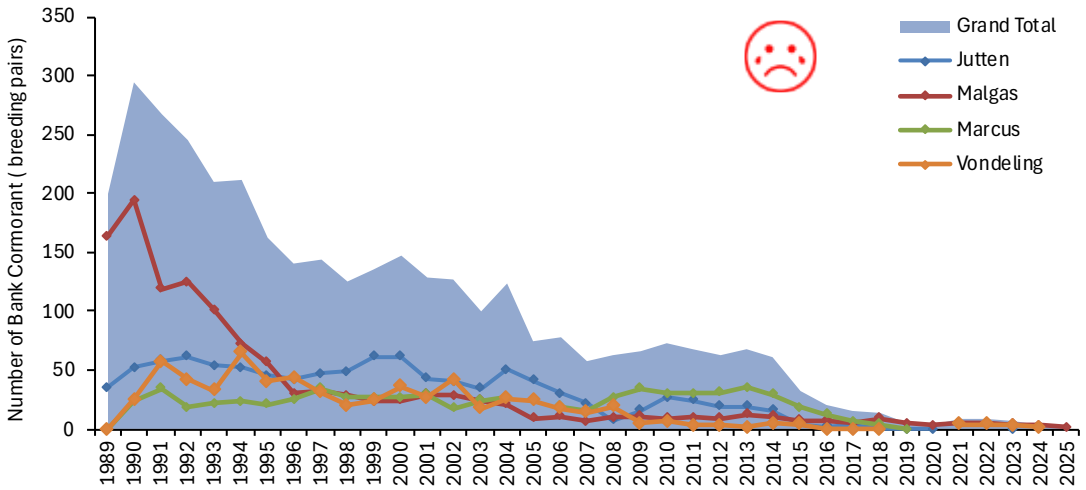


Figure 10.9. Trends in breeding population of Bank Cormorants at Jutten, Malgas, Marcus and Vondeling Islands in Saldanha Bay from 1980–2025 measured in number of breeding pairs. Data source: Seakamela at al. (2026).

During Pelican Watch monitoring in the 2024–2025 breeding season, just three Bank Cormorant nests were observed on Malgas Island, down from seven nests the previous season. Overall, the population in Saldanha Bay has declined drastically by approximately 99% since 1990 (Figure 10.9). DFFE survey data reveal that the numbers of breeding pairs in the last five years are the lowest on record, with between just one and nine pairs recorded (Figure 10.9). Sadly, it appears that soon no Bank Cormorants may breed on the Saldanha islands and the colony will be lost. 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 (Sherley et al. 2012, BirdLife International 2018c).

The **White-breasted Cormorant** *Phalacrocorax carbo,* also known as Great Cormorant, occurs along the entire southern African coastline, and is common in the eastern and southern interior 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 Mugilid fishes such as steentjies, white stumpnose and harders (BirdLife International 2023). 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 1920s, 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 since 2004 has been on Meeuw Island (Figure 10.10). Overall, numbers of breeding pairs were more or less stable until 2012 but have declined steeply thereafter. The annual counts made between 2015–2025 were substantially down from the 100–150 breeding pairs recorded in most years prior to 2012 (Figure 10.10).

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 (BirdLife International 2023). 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 and the average size of harders in the Saldanha- Langebaan fishery has decreased (see Chapter 9: Fish and economically important invertebrates), potentially negatively affecting foraging opportunities for White-breasted Cormorants in the Bay. Due to Schaapen Island's 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 two decades could have been a contributing factor to the shift in breeding location from Schaapen to Meeuw Island in 2004, but this appears unlikely given that the opposite shift happened ten years previously.

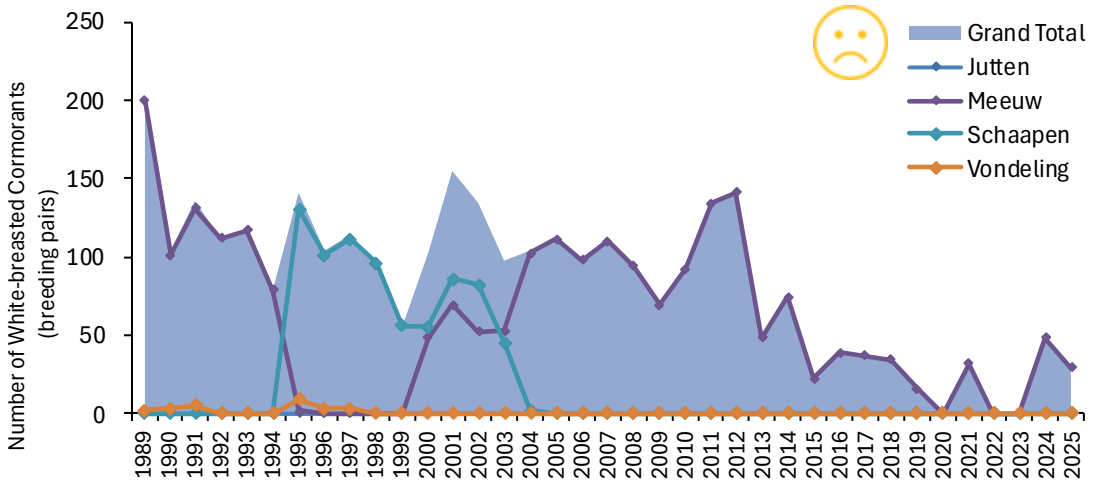


Figure 10.10. Trends in breeding population of White-breasted Cormorants at Jutten, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1980–2025 measured in number of breeding pairs Data source: Seakamela at al. (2026).

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 Least Concern on the IUCN Red List and despite being small (~4 300 individuals) and range restricted, its population trend is stable (IUCN 2025). This species is thought to be highly susceptible to human disturbance, but the Saldanha Bay population does not appear to be unduly affected by Port and mariculture operations. It is susceptible to predation by fur seals, particularly of fledglings (David et al. 2003). 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 (BirdLife International 2023). Populations of this species have been comprehensively counted since 1991 (Figure 10.11). Since then, numbers have shown considerable interannual variations with an overall decreasing trend (Figure 10.11**).** Currently, numbers are well below average, and the third lowest in the last three decades. Furthermore, the trajectory in population size has been downwards for the last decade. In recent years Crowned Cormorants have started breeding under the Iron Ore Terminal in Saldanha Bay where at least 30 pairs were observed nesting every year since 2023. This shift in breeding location may account for some of the decline in the number of pairs breeding on the islands. Crowned Cormorants are also commonly encountered birds roosting on mariculture infrastructure in Small Bay, with the average count over the period March 2024–May 2026 of 110 birds (Figure 10.1).

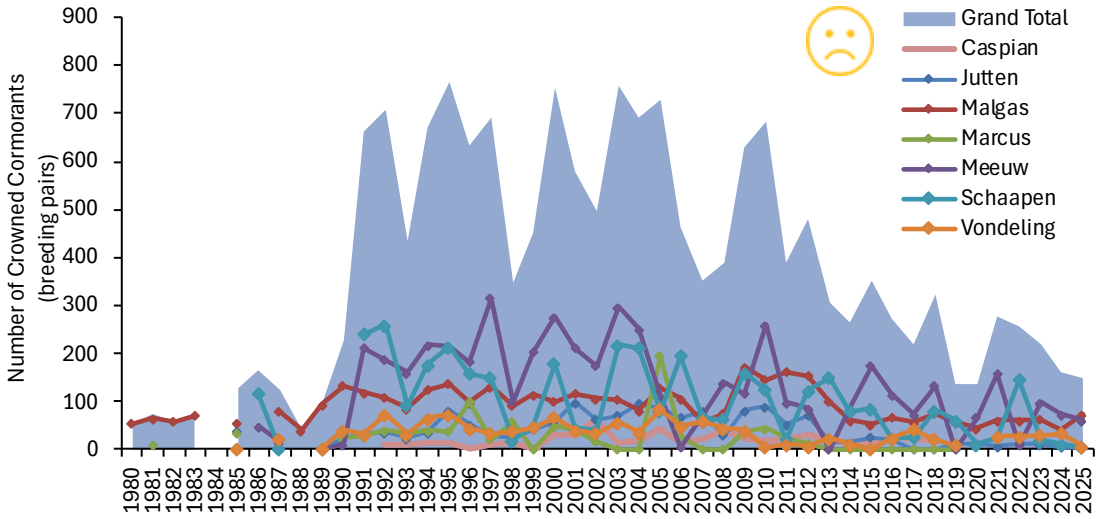


Figure 10.11. Trends in breeding population of Crowned Cormorants at the Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling, and Caspian Islands in Saldanha Bay from 1980–2025 measured in number of breeding pairs. Data source: Seakamela at al. (2026).

10.2 BIRDS OF LANGEBAAN LAGOON

10.2.1 NATIONAL IMPORTANCE

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,* Red Knot *Calidris canutus* 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 in the summer of 1976–77, where 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.

The Coordinated Waterbird Counts (CWAC) project was started in 1992 by the Animal Demography Unit and the data which extend back to the 1970s are curated by the University of Cape Town's FitzPatrick Institute of African Ornithology. The CWAC database contains summer and winter counts from Langebaan Lagoon for most years since 1975, except for 2020, 2023 and 2024 when no winter counts were undertaken.

Waterbird numbers on Langebaan Lagoon have declined dramatically since monitoring began in the 1970s. Decreases in both migratory and resident wader numbers are a common trend globally and around the South African coast (Zöckler et al. 2003, Essig 2016), and can be attributed to loss of breeding habitat and hunting along their migration routes, human disturbance and habitat loss on their wintering grounds, climate change, pollution and other forms of direct human disturbance (Zöckler et al. 2003). 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; WCNP), and recreational activities such as boating, angling and swimming have since been controlled within the Lagoon through zonation. Nevertheless, the dramatic increases in visitor numbers to the area over the last three decades and the more recent increases in sporting activities impact on some of the important feeding areas in the lagoon. The area most impacted by

increased visitor numbers and water sports activities are the sandflats near Oesterwal (Shark Bay), that in the 1970s were identified as one of the most important feeding areas for waders.

10.2.2 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 10.1). 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 10.12). On average, over the last six years, waders account for about 73% 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 9% to 55% of total bird numbers (Figure 10.12). The influx of waders into the area during summer accounts for most of the seasonal change in community composition. Most of the Palaearctic migrants depart synchronously in early April, but the immature birds of many of these species remain behind, accounting approximately 13% of the total waterbird numbers. The resident species take advantage of relief in competition for resources and use this period to breed. The migrants return over a longer period in spring, with birds beginning to filter in from August, rising rapidly in numbers during September and November. In the 1970s, it was determined that the most important sandflats, in terms of the density of waders they support, were in Rietbaai, in the upper section of Langebaan Lagoon, and at the mouth, near Oesterwal. The important roosting sites were the salt marshes, particularly between Bottelary and Geelbek (Summers 1977).

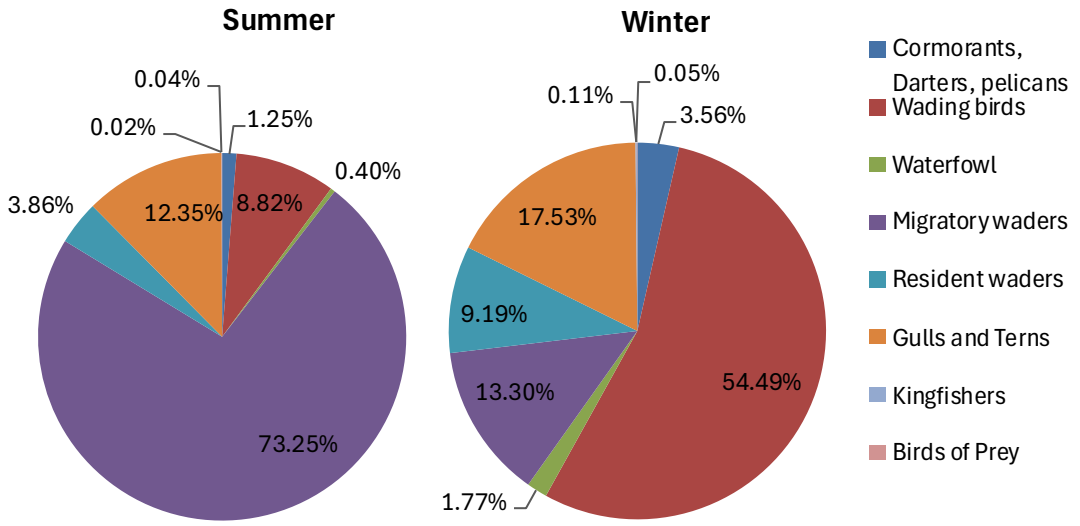


Figure 10.12. The current average numerical composition of the waterbirds on Langebaan Lagoon during summer (2019–2026) and winter (2017–2025) (Data source: GBIF.org (5 June 2026), CWAC data, Animal Demography Unit at the University of Cape Town).

Table 10.1. 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 Shoveler. 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, plovers). 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.
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.

10.2.3 TRENDS IN BIRD NUMBERS

Declines in waterbird numbers in Langebaan Lagoon is largely due to changes in the numbers of waders, which used to account for more than 90% of the total water bird numbers (Figure 10.13). Migratory wader bird numbers have since decreased significantly and fluctuated around 7 000 (range: 2 352–12 940) individuals in summer

counts conducted over the period 2011–2026 (Figure 10.13). The 12 940 migratory waders counted in the summer of 2021 is the highest in the last decade. The reasons for this increase are not known but it is postulated that reduced human activity throughout their range due to global COVID-19 lockdowns in 2020, may have facilitated the modest recovery (still only about 40% of the pre-1990 average of ~33 000 birds). Unfortunately, the counts of migratory waders conducted during the summers of 2022–2026 were all about half that recorded in the summer of 2021, and it appears that the "recovery", was only temporary. Note that the increase in wading bird counts in 2021 probably did not represent a demographic recovery i.e., a population increase, as the timeframe of COVID-19 lockdowns was probably too short for such a response. However, some reduction in human caused mortality may have occurred e.g., a short-term decline in hunting of waders during their migration over southern Europe and North Africa (Barbosa 2001). It does, however, show a local increased use of the Langebaan habitat during the summer of 2020–2021, possibly in response to decreased human activity. Overall, the sustained decline in migratory waders between about 1990 and 2010 appear to have stopped with numbers stabilizing since then, albeit at about a quarter of historical levels) (Figure 10.13). Resident wader numbers also appear to have stabilised since 2010 at around 50% of the pre-1990 levels, suggesting that management measures such as zonation have been effective in halting the declines.

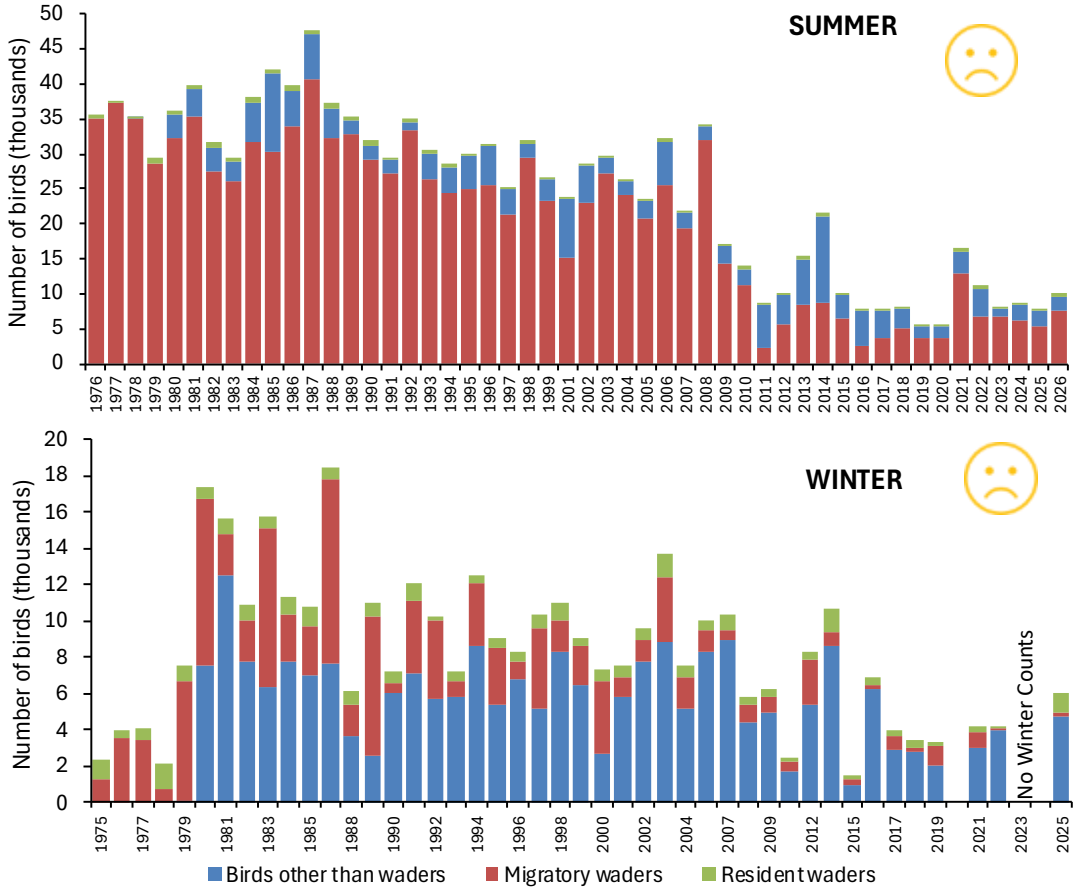


Figure 10.13. Long-term trend in the numerical composition of waterbirds in the Langebaan Lagoon during summer (top) and winter (bottom) (1975–summer 2026). Note that no data were collected in the summer of 1975, as well as in the winter of 1987, 2006, 2010, 2014, 2020 & 2023–2024 (Data source: GBIF.org (5 June 2026), Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2025).

Drastic population declines in four species, including the Ruddy Turnstone, Red Knot, Grey Plover, and Curlew Sandpiper typify this downward trend in summer migratory bird numbers. Most importantly, Curlew Sandpiper numbers dropped from a pre-1990 average of just over 20 000 birds to a minimum in 2011 with only 413 individuals. Curlew Sandpiper numbers have not recovered since, with an average annual count of ~2 800 individuals over the last decade. Prior to 1990, this species accounted for almost two thirds of the total summer migratory wader numbers in the Lagoon. The overall recovery in migrant wader numbers in 2021 was also not seen in all species, for example Curlew Sandpiper, Ruddy Turnstone and Grey Plover counts were recovered to a similar level (approximately 30–40% of pre-1990 numbers), whilst Red Knot numbers showed almost no increase and remained well below their historical peak. The most recent 2026 summer count reveals how short-lived this recovery was with all four species of the formerly most abundant migrants (Red Knot, Curlew Sandpiper, Grey Plover and Ruddy Turnstone), at a small fraction (0–20%) of their pre-1990 average abundance. Migratory waders that have not seen such drastic declines and were encountered in historical numbers in 2026 include Common Ringed Plover (311% of pre-1990 average in 2026), Common Greenshank (55%), Common Whimbrel (66%) and Little Stint (211%). Counts of these species seldom numbered more than a thousand birds historically, and interannual variability in the counts has always been high, but it is reassuring that in the summers of 2024–2026, counts were similar to or higher than historical counts and the low numbers of these species recorded in the 2023 summer were outliers. Overall, however, due to the large declines in the numbers of historically most abundant species, migratory wader numbers from the summer of 2026 were just 23% of the historical (pre-1990) average (Figure 10.13).

Resident wader numbers have also fluctuated widely over time, but numbers appear to have been on a negative trajectory since 2007, with no signs of sustained recovery in the last decade. The winter 2019 count was the lowest on record and unfortunately there was no count for the winter of 2020 (Figure 10.13). The 2021 winter count of resident waders (338) was similar to the average in recent years, which suggested that if these residents had benefited from reduced disturbance during 2020 COVID-19 lockdowns, it was unfortunately very short-lived. The poor status of resident wader populations in Langebaan Lagoon was reflected in the all-time low winter count of just 142 birds recorded August 2022. No winter counts have been undertaken in 2023 or 2024, but the 2025 winter count was the second highest in the last two decades, suggesting some improvements in the status of resident waders. There, has, however, always been high variability in winter resident wader counts with a standard deviation of 317 (~50%) of the mean of 644 birds over the nearly 50-year data record. Average winter counts of resident waders over the last 15 years have been about half the pre-1990 average (Figure 10.13).

Summer resident wader counts show a similar short-term recovery attributed to decreased disturbance during COVID-19 lockdowns with the summer 2021 and 2022 resident wader counts of 605 and 543 respectively being similar to the pre-1990 average of 623. Thereafter, the average summer resident wader count for 2023–2024 was just 188 birds, less than a third of the pre-1990 average, i.e. the short-term increase in resident wader numbers in 2021 and 2022 was not sustained and was attributed to decreased disturbance that resulted in increased use of the lagoon habitats, rather than a sustained population recovery. The 2026 summer resident wader count was, however, encouraging, with a 20-year high of almost 700 birds recorded including high than average numbers of Oystercatchers, Black winged Stilts, Pied Avocets and Kittlitz's Plovers.

In conclusion, the reasons for the long-term declines, particularly in migratory wader numbers, are 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 (Zöckler et al. 2003, Dias et al. 2006). Hunting of migratory waterbirds is a strong tradition in several European and North African countries and is thought to contribute towards global declines in migratory water birds (Barbosa 2001, Zöckler et al. 2003). Ryan (2013) 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 suggests that factors outside of the Western Cape were at least partially responsible for the observed trends and probably reflected global population declines (Ryan 2013). Conditions at Langebaan Lagoon could also have contributed to the decline in wader numbers over the two decades between 1990–2010. 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 12: Aquatic Macrophytes), and human disturbance, which has been shown to have a dramatic impact on bird numbers in other estuaries (Turpie & Love 2000). Some historically important wader feeding areas such as Oesterwal, lie within the zones that are highly utilised for recreation. However, as mentioned above, wader numbers in Langebaan appear to have stabilised since approximately 2011, suggesting conditions are no longer deteriorating and that management measures have arrested the decline.

10.3 Cape Fur Seals

The Cape fur seal *Arctocephalus pusillus pusillus* is the only seal species that breeds in Southern Africa. Its range extends from the centre of Angola to the east coast of South Africa, with breeding colonies extending south from Baia dos Tigres on the southern border of Angola, through Namibia, down the west coast of South Africa and around to Algoa Bay in the Eastern Cape of South Africa (Figure 10.14). Before 1900, it is likely that seals were present on most (if not all) islands off South Africa and Namibia, where they prefer to breed as they are protected from mainland predators. However, populations on many of the islands were significantly depleted or disappeared completely as a result of uncontrolled hunting, and human occupation of the islands for the collection of guano and other seabird related products (Kemper et al. 2007). Subsequent to the ban on seal hunting, the Cape fur seal population recovered, showing an almost 20-fold growth in numbers in the 20th century before stabilizing at about two million animals (Kemper et al. 2007). In addition, the number of breeding colonies have increased since 1970 from 23 to 40 colonies (Kirkman et al. 2013).

The overall population count has reportedly remained largely unchanged since 1993 and is estimated at 1.5–2.0 million, although the distribution of these seals has been shown to vary in relation to prey distribution and shortages (Kemper et al. 2007, Kirkman et al. 2013, 2019).

Rabies has recently (June 2024) been detected within the South African seal population (DLRRD 2024). Testing of samples dating back to August 2022 revealed that rabies was already present in seals and the detection of the virus in seals from Melkbostrand to Plettenberg Bay suggests that it has been circulating in the population for some time and is now endemic (DLRRD 2024). Sequencing of positive seal samples revealed that it is a canid rabies strain with the probable origin being a black-backed jackal in Namibia.

Despite several attacks by likely rabid Cape fur seals on humans in the last 5–6 years, no cases of seal to human rabies transmission have been recorded. However, it is thought that a seal to dog transmission has taken place indicating that rabies transmission from seals is possible (DLRRD 2024). To date rabies does not appear to be causing high number mortalities amongst Cape fur seal populations–in other words, it is not having a detectable demographic effect. Any seal behaving abnormally or aggressively should be avoided and if a human or their pet is bitten, immediate medical or veterinary care is required.

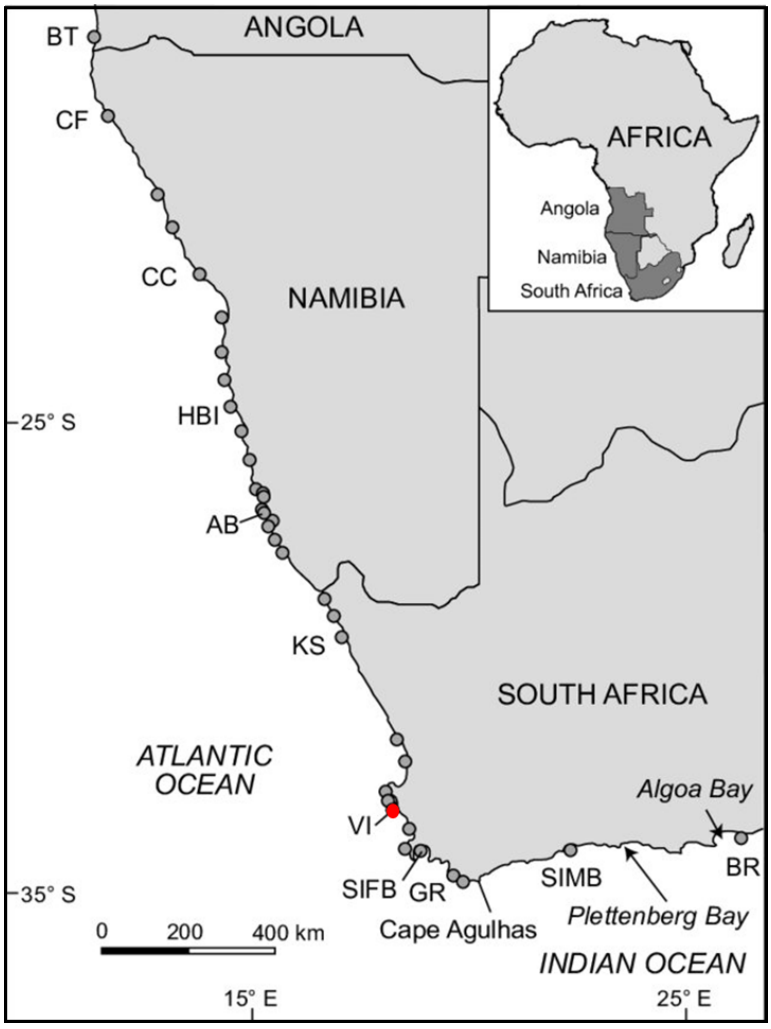


Figure 10.14 Distribution of Cape fur seal breeding colonies of Southern Africa from Black Rocks (BR), Algoa Bay to Baia dos Tigres (BT), southern Angola, with selected breeding colonies indicated including Seal Island in Mossel Bay (SIMB), Geyser Rock (GR), Seal Island in False Bay (SIFB), Vondeling Island (VI), Kleinsee (KS), Atlas Bay (AB), Hollam's Bird Island (HBI), Cape Cross (CC) and Cape Frio (CF). The red star indicates the location of the recently established breeding colony at Vondeling Island just outside Saldanha Bay. Adapted from Huisamen et al. (2011).

Seal populations along the west coast of South Africa exhibited a fairly stable distribution over the period 1976 to 2006, with the centre of the distribution of the breeding colonies remaining fixed within the region (Kirkman et al. 2013). However, more recently, it has been noted that there is a general shift southwards of St Helena Bay with new breeding colonies being established on Vondeling Island, at Cape Point and potentially on the

South Coast near Betty's Bay (DFFE, Mduduzi Seakamela *pers. comm.* 2020). It is possible that this southward shift coincides with a corresponding eastwards shift of small pelagic fish species which are a key food source for the seals (see Figure 10.3). Although seals historically would frequent the seabird islands (Jutten, Malgas, Marcus and Vondeling) around Saldanha Bay, coming on land (hauling out) to rest or sun themselves (David & Van Sittert 2008), it is only since the turn of the century that a breeding colony has been established on Vondeling Island, south of the entrance to Saldanha Bay (Seakamela et al. 2022).

The DFFE monitors seal populations at 11 colonies in South Africa through aerial surveys which are undertaken to count pup numbers and hence to track seal population trends over time. As it is a newly established breeding colony, aerial surveys have been conducted at Vondeling Island every year since 2006 (DFFE, Mduduzi Seakamela *pers. comm.* 2020), with exceptions to this being in 2007, 2016, 2021,2022, and 2024) when counts were not undertaken as a result of logistical constraints (Seakamela et al. 2026). Initially, the number of pups on the island increased dramatically up until 2010, after which the rate of increase slowed and pup numbers on the island have fluctuated at between 17 000–23 400 pups over the period 2014–2024 (Figure 10.15). These fluctuations suggest that the island may have reached carrying capacity, with the annual changes linked to the availability of prey resources and increases and decreases in the colony size mirroring those of sardines and anchovies on the West Coast (Figure 10.15). Possibly in response to improved sardine biomass on the west coast, seal pup numbers on Vondeling Island in 2025 increased to the highest on record with nearly 27 000 counted. The PHV outbreak and associated sardine stock declines may well result in decreased seal pup counts in the coming breeding seasons.

Cape fur seals are amongst the largest marine top predators found in and around Saldanha Bay. They are opportunistic, generalist feeders that have been shown to benefit from human activities including utilisation of discards from fishing boats, or taking fish directly from fisherman (Wickens et al. 1992, Makhado et al. 2009). In addition, seals compete with seabirds, such as penguins and gannets, as well as with commercial fisheries, for small pelagic fish, which form a key part of their diets (Crawford et al. 2011, de Moor & Butterworth 2015). Kirkman et al. (2013) suggested that the increasing numbers of seals on Vondeling Island may lead to increased pressure to cull seals both from a fisheries perspective as well as to protect important seabird species on which seals are known to prey. In fact, some culling has been undertaken of seals off the West Coast in recent years in an attempt to limit the mortality of seabirds that are of conservation importance. The culling of 'problem' seals seen killing Cape Gannet fledglings at Malgas Island, located northwest of the Vondeling breeding colony, resulted in a reduced mortality of gannet fledglings, however, seals learnt to avoid the boat used for culling, and the predation of seabirds around the island is ongoing (Makhado et al. 2009).

Studies investigating the predation of seabirds by seals at Malgas Island outside Saldanha Bay show that the 'problem' seals are restricted to sub-adult males which are generally less than five years old (Makhado et al. 2006, 2009). These individuals are not confined to the breeding colonies and are too young to be part of a breeding harem and therefore tend to be more nomadic with inconsistent feeding areas. Data collected by DFFE using GPS trackers attached to female Cape fur seals tagged on Vondeling Island, indicates that these animals favour offshore feeding grounds and do not enter Saldanha

Bay at all (Figure 10.16 and Mduduzi Seakamela *pers. comm.* 2020). It is likely that in order to maintain sufficient body fat and health to produce enough milk to support their pups, these females prefer the high-quality food provided by the small pelagic fish species as opposed to irregular and limited food sources associated with fledgling birds. In addition, the females are more likely to be disturbed by human activities in the Bay, and thus elect to avoid contact with humans to reduce the risk of conflict and therefore the risk of not returning to the breeding colony and their pups. This is supported by research showing that breeding and pupping harbour seals on the west coast of North America have been displaced by shellfish aquaculture activities (Becker et al. 2011). However, Cape Fur Seals are attracted to the aquaculture infrastructure within Saldanha Bay, in particular the rafts that they use extensively as haul outs for resting (Figure 10.1). Over the preceding decade it appeared that the carrying capacity of Vondeling Island had been reached. It remains to be seen if the 2025 record of nearly 27 000 pups was just a fluctuation around carrying capacity or if numbers will continue to increase in coming breeding seasons.

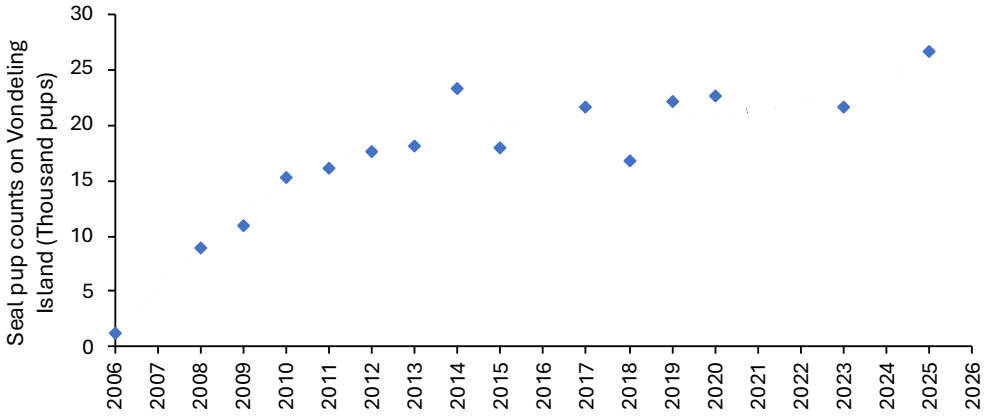


Figure 10.15. Trends in seal pup counts collected during aerial surveys conducted at Vondeling Island, Saldanha Bay from 2006–2025. No data available for 2007, 2016, 2021, 2022 & 2024. Source: Seakamela at al. (2026).

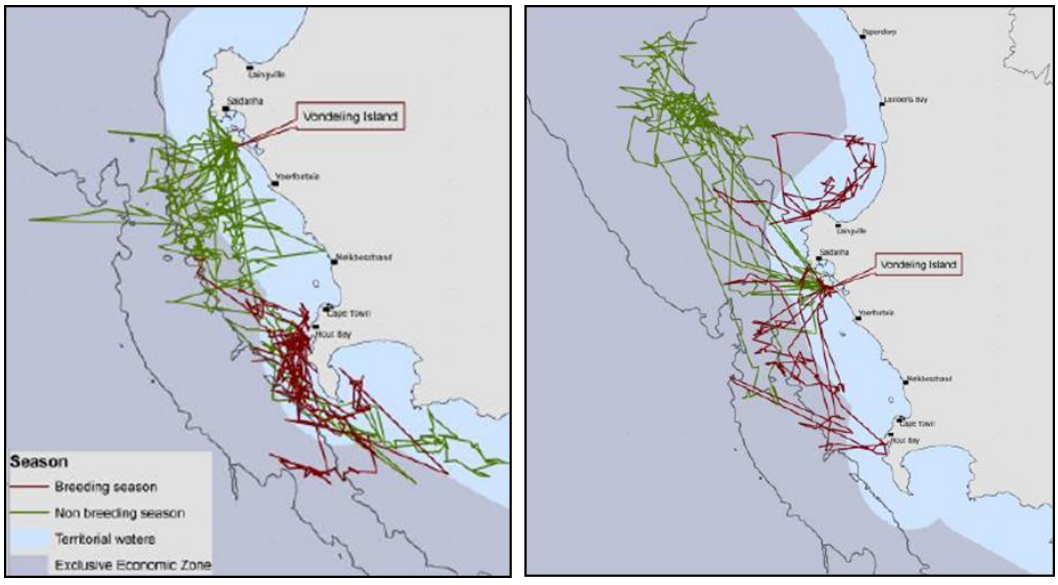


Figure 10.16. GPS tracks of female Cape fur seals tagged on Vondeling Island showing the routes travelled during the breeding and non-breeding season (Source: DFFE: Oceans and Coasts).

11 ALIEN AND INVASIVE SPECIES

11.1 BACKGROUND

Human-induced biological invasions have become a major cause for concern worldwide. The life history characteristics, ecological resilience of the affected areas, presence of suitable predators, biotic resistance and propagule pressure are some of the many factors that influence whether an alien species becomes a successful invader or not. Biological invasions can have a negative impact on biodiversity and may result in local, or even the global, extinction of indigenous species. Alien species invasions can also have tangible and quantifiable socio-economic impacts, and management of these species is vital. A precautionary approach to prevent biological invasions is often considered the most efficient method of management and can include identifying and managing important pathways of introduction. Regular monitoring is imperative to enable the early detection of alien species, guide rapid responses and support targeted control and management measures before populations proliferate, thereby reducing the impacts of these invaders on the receiving environment and biota as far as possible.

The revised, internationally accepted standard recognises an alien species as invasive if it has self-replacing populations over several generations, and has expanded its range beyond the point of introduction (Wilson et al. 2009, Richardson et al. 2011, Blackburn et al. 2011). This approach has been proposed for South African marine invasion biology research (Robinson et al. 2016).

The most recent published review of marine alien species in South Africa identified a total of 96 alien species, with 55 classified as invasive (Robinson et al. 2020). However, given ongoing discoveries and evolving species statuses, the list of alien species in South Africa is dynamic, and at the time of drafting, 145 introduced marine species have been recorded from South Africa, with 64 classified as invasive, 38 as cryptogenic, 37 as alien, and six as 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) (Tammy Robinson-Smythe *pers. comm.* June 2026). This introduced fauna is largely comprised of the phyla Arthropoda (amphipods, isopods, crabs, barnacles), Mollusca (bivalves, gastropods), and Chordata (ascidians or sea squirts) (Figure 11.1). Shipping is a major vector, with 95% of these species introduced via activities such as ballast water discharge and hull fouling (Robinson et al. 2020). Hard artificial substrata within ports also facilitate the introduction, establishment and regional spread of alien species (Outinen et al. 2021, Janiak & Branson 2021). As such, harbours are one of the most invaded habitats of the marine realm, and comprise the highest concentration of marine alien species in South Africa (Picker & Griffiths 2017, Leclerc & Viard 2018).

11.2 APPROACH FOR MONITORING

Sampling and monitoring of alien species within Saldanha Bay forms an important part of the State of the Bay (SOB) monitoring programme. Quantitative data are obtained from the Benthic Systems (Chapter 8) and Rocky Shore monitoring surveys for three invasive species, namely the Western pea crab *Rathbunixa occidentalis*, Pacific barnacle *Balanus glandula*, and mytilid mussel species complex. In addition, qualitative data are collected

when conducting any other surveys in the Bay. This has previously included Aquaculture Development Zone (ADZ) biofouling surveys for the Department of Forestry, Fisheries and the Environment in 2021 and 2022 (Payne et al. 2022, 2023b), as well as eDNA-based surveys (in collaboration with NatureMetrics) for Anglo Shipping in 2022 (Payne et al. 2023a), 2025 and 2026.

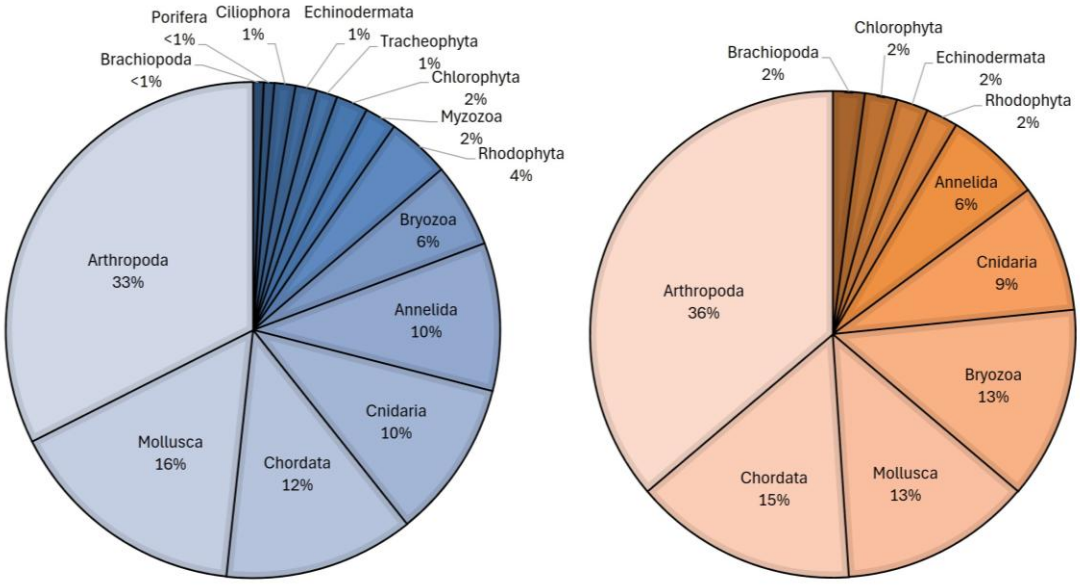


Figure 11.1. Phyla composition of marine introduced species recorded in South Africa (left) and alien species recorded in Saldanha Bay and Langebaan Lagoon (right).

11.3 CONFIRMED ALIEN AND INVASIVE SPECIES

To date, 47 alien species have been recorded in Saldanha Bay and Langebaan Lagoon, 40 of which are considered invasive, and two naturalised (Table 11.1). This alien fauna is largely comprised of the phyla Arthropoda (amphipods, crabs, barnacles), Chordata (ascidians or sea squirts), Mollusca (bivalves, gastropods), and Bryozoa (moss animals) (Figure 11.1). A synopsis of the state of knowledge of these alien species is provided below.

11.3.1 SHELL WORM *BOCCARDIA PROBOSCIDEA*

A spionid polychaete of yellow-orange colouration, with red branchiae (gills), conspicuous segmentation and thick anterior tentacle-like palps (Figure 11.2 A). It has a long, somewhat flattened, body that tapers towards the rear (Piazolla & Hiebert 2015). In South Africa, this species is known to occur at a number of oyster and abalone farms, and has been recorded in Saldanha Bay (Haupt et al. 2010). This species creates shallow, sand-lined burrows on the surface of oysters, abalone and other shellfish. Due to its broad thermal tolerance and flexible reproductive strategy, *B. proboscidea* has the potential to disperse and establish itself along the South African coastline, although it is believed that anthropogenic movement will be the primary factor governing its spread and establishment (David et al. 2016).

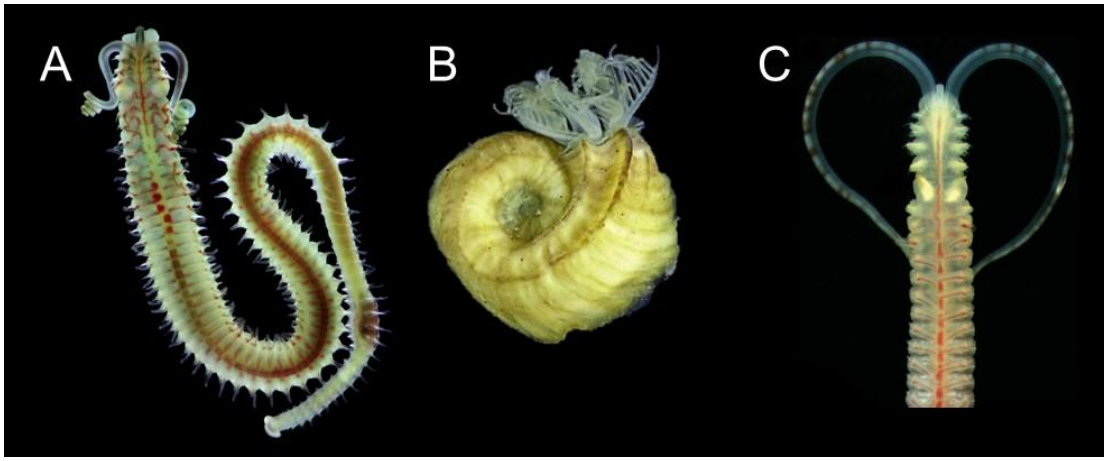


Figure 11.2. Alien invasive polychaetes recorded from Saldanha Bay and/or Langebaan Lagoon. A) shell worm *Boccardia proboscidea* (Leslie Harris), B) *Neodexiospira brasiliensis* (Teresa Darbyshire), C) *Polydora hoplura* (Radashevsky et al. 2017).

11.3.2 POLYCHAETE *NEODEXIOSPIRA BRASILIENSIS*

A wide-ranging, small, epifaunal polychaete that grows in tightly coiled, calcareous tubes (Fofonoff et al. 2019) (Figure 11.2 B). It was first collected in South Africa in 1953 (Mead et al. 2011a), and is known to occur from Saldanha Bay to Durban (Mead et al. 2011a, Loureiro et al. 2021). This species can cause eelgrass blade leaf damage in some areas where it has been introduced (Fofonoff et al. 2019).

11.3.3 POLYCHAETE *POLYDORA HOPLURA*

A large worm, attaining 6 cm in length and 2 mm in width with more than 200 chaetigers (chaetae-bearing segments) (Radashevsky et al. 2017) (Figure 11.2 C). It bores into calcareous-shelled organisms and is commonly found in abalone and oyster mariculture facilities (Fofonoff et al. 2019). This polychaete was first recorded in South Africa in 1947 from Table Bay (Millard 1951), and was subsequently reported from the intertidal and shallow waters of Saldanha Bay to Plettenberg Bay. This species commonly infests the commercially cultured oysters (Day 1967, Nel et al. 1996) and abalone in Saldanha Bay (Simon et al. 2006, Boonzaaier et al. 2014), which can have negative economic implications as it decreases their condition and/or survival (Fofonoff et al. 2019).

11.3.4 JAPANESE SKELETON SHRIMP *CAPRELLA MUTICA*

A large caprellid amphipod, orange to red in colour (Figure 11.3 A). Males can be up to 3.5 cm long, and females 1.5 cm, with the brood pouch covered in dark red spots (Oakley 2006). It was first recorded in South Africa from False Bay Marina (Peters & Robinson 2017), in which it has reached substantial densities and is of concern due to its fast reproductive cycle and high reproductive output. Its detection in the 2022 DNA-based survey represented the first record of this species in Saldanha Bay, and the second record in South Africa (Payne et al. 2023a). It has since been detected in Saldanha Bay in 2026 via eDNA metabarcoding.

Table 11.1 Marine alien (A), Naturalised (N), and invasive (I) species recorded in Saldanha Bay and Langebaan Lagoon from previous Anchor surveys (2006–2026), and other published studies. Vectors include ship boring (SB), ship fouling (SF), ballast water (BW), solid ballast (BS), oil rigs (OR) and mariculture (M). DNA based detections are from eDNA surveys (in collaboration with NatureMetrics) undertaken for Anglo Shipping in 2022 (Payne et al. 2023a), 2025 and 2026.

Class	Species	Common name	Status	Natural Range	Vector	Anchor surveys				Other Research
						Benthic Macro	Rocky Shore	ADZ Biofouling	DNA	
Kingdom Animalia										
Phylum Annelida										
Polychaeta	*Boccardia proboscidea*	Shell worm	I	North Pacific	M				X	X
	Neodexiospira brasiliensis	-	I	West Indies, Brazil	SF/BW					X
	Polydora hoplura	-	I	Europe, Mediterranean	SF/BW/M				X	X
Phylum Arthropoda										
Malacostraca	*Caprella mutica*	Japanese skeleton shrimp	I	Japan	SF/BW			X	X	
	Carcinus maenas	European shore-crab	I	Europe, Mediterranean	SF/BW/OR					X
	Cerapus tubularis	Tubular amphipod	I	-	SF/BW					X
	Chelura terebrans	Wood-boring amphipod	I	North Atlantic	SF/SB					X
	Ericthonius brasiliensis	-	I	North Atlantic	SF/BW	X				
	Homalaspis plana	Chilean stone crab	A	Chile	SF					X
	Jassa slatteryi	-	I	Pacific North America	SF/BW					X
	Monocorophium acherusicum	Fat-feeler amphipod	A	North Atlantic	SF/BW	X			X	X
	Orchestia gammarellus	Shore-hopper	I	Europe, Mediterranean	BS					X
	Platorchestia platensis	Beach-flea	I	-	BS					X
	Porcellana africana	Porcelain crab	I	North-west Africa	SF/BW		X	X	X	X
	Pyromaia tuberculata	Tuberculate pear crab	I	Pacific North America	SF/BW	X			X	
	Rathbunixa occidentalis	Western pea crab	I	Pacific North America	SF/BW	X				X
Thecostraca	*Amphibalanus venustus*	Striped barnacle	I	Tropical North Atlantic	SF					X
	Balanus glandula	Pacific barnacle	I	Pacific North America	SF/BW		X		X	X
	Balanus trigonus	Pitted/triangle barnacle	I	Pacific	SF/BW				X	X
	Perforatus perforatus	Perforated barnacle	A	Atlantic Ocean	SF/BW		X		X	X

Class	Species	Common name	Status	Natural Range	Vector	Anchor surveys				Other Research
						Benthic Macro	Rocky Shore	ADZ Biofouling	DNA	
Phylum Brachiopoda										
Lingulata	*Discinisca tenuis*	Disc lamp shell	I	Namibian Coast	M			X	X	X
Phylum Bryozoa										
Gymnolaemata	*Amathia verticillata*	Spaghetti bryozoan	I	Mediterranean	SF					X
	Bugula neritina	Fouling moss animal	I	-	SF				X	X
	Bugulina flabellata	Fan-shaped moss animal	I	-	SF					X
	Conopeum seurati	-	I	Europe	SF					X
	Cryptosula pallasiana	Red crust	I	Europe	SF					X
	Watersipora subtorquata	Red-rust bryozoan	I	Caribbean	SF		X		X	X
Phylum Chordata										
Ascidiacea	*Ascidiella aspersa*	Dirty sea squirt	I	North Sea	SF					X
	Asterocarpa humilis	-	I	-	SF					X
	Botryllus schlosseri	Golden star ascidian	I	Northeast Atlantic	SF				X	X
	Ciona robusta	Sea vase	I	Europe	SF			X	X	X
	Clavelina lepadiformis	Bell ascidian	I	Europe	SF				X	X
	Diplosoma listerianum	Gossamer ascidian	I	Europe	SF					X
	Styela plicata	Pleated sea squirt	I	West Pacific	SF					X
Phylum Cnidaria										
Hexacorallia	*Cereus pedunculatus*	European daisy anemone	N	Europe, Mediterranean	BW/M					X
Hydrozoa	*Coryne eximia*	-	I	North Atlantic, Pacific	SF/BW					X
	Ectopleura crocea	Pinkmouth hydroid	A	North Atlantic	SF/BW			X	X	
	Obelia dichotoma	Thin-walled obelia	N	-	SF/BW				X	X
Phylum Echinodermata										
Asteroidea	*Heliaster helianthus*	South American sunstar	A	South America	SF					X
Phylum Mollusca										
Bivalvia	*Magallana gigas*	Pacific oyster	I	Japan, Northwest Pacific	M					X

Class	Species	Common name	Status	Natural Range	Vector	Anchor surveys				Other Research
						Benthic Macro	Rocky Shore	ADZ Biofouling	DNA	
	Mytilus edulis	Blue mussel	I	North Atlantic	SF/BW/M				X	
	Mytilus galloprovincialis	Mediterranean mussel	I	Mediterranean, Northeast Atlantic	SF/BW	X	X	X	X	X
	Mytilus trossulus	Bay mussel	I	North Pacific	SF/BW/M				X	
	Semimytilus patagonicus	Bisexual mussel	I	Pacific South America	SF/BW				X	X
Gastropoda	*Littorina saxatilis*	British periwinkle	I	Europe, Mediterranean, West Atlantic	BS					X
Kingdom Plantae										
Phylum Chlorophyta										
Ulvophyceae	*Codium fragile*	Fragile upright codium	I	Korea	SF		X			X
Phylum Rhodophyta										
Florideophyceae	*Antithamnionella spirographidis*	-	I	-	SF/BW/M					X

11.3.5 European shore-crab *Carcinus maenas*

A mottled khaki-green crab with oval carapace, and five marginal teeth on each side (Figure 11.3 B). The legs are robust with flattened but pointed tips (Branch et al. 2022). In South Africa, this species was first collected from Table Bay Docks in 1983, and has since established dense populations, consequently decimating shellfish populations (Robinson et al. 2005a). This species is typically restricted to sheltered, coastal sites and appears thus far unable to establish on the open wave-swept coastline of South Africa (Hampton & Griffiths 2007). Surveys in Saldanha Bay have not detected any living individuals, but a single dead specimen was recorded in the Small Bay Small Craft Harbour (Robinson et al. 2004b). More recently, a single specimen was recorded from Churchhaven (Alison Kock *pers. comm.* August 2025). Nevertheless, it is unlikely that there exists an extant population in Saldanha Bay.

11.3.6 Tubular amphipod *Cerapus tubularis*

A small and slender amphipod that lives in a characteristic tapering tube (Branch et al. 2022) (Figure 11.3 C). It was first recorded from South Africa off the coast of KwaZulu-Natal in 1901, but was incorrectly reported as *Cerapus abditus* (Barnard 1916). This species is currently known to occur from Saldanha Bay to the east coast of South Africa (Mead et al. 2011a).

11.3.7 Wood-boring amphipod *Chelura terebrans*

A marine amphipod with a body length of 3.5–6 mm (Figure 11.3 D). It is depressed and cylindrical, with a broad and rounded head, and a pale brown colour, mottled with pink (Fofonoff et al. 2019). Some of the first specimens in South Africa were collected in 1888 (Stebbing 1910). More recently, it has been reported in all harbours between Langebaan and Gqeberha, although further surveys are required to determine if it has spread to the open coast (Mead et al. 2011a). It was likely introduced by hull fouling of the wooden ships used historically (Kuhne & Becker 1964). A wood-boring amphipod, it is dependent upon wood-boring isopods of the genus *Limnoria* for shelter and food as it inhabits their burrows and feeds on their faecal matter (Kuhne & Becker 1964, Borges et al. 2010). Its diet also includes bacteria, protists, and decaying wood. Due to its wood-boring nature, it is considered a pest, but only in the presence of *Limnoria*. This species thus has the potential to negatively impact the economy by destroying wooden structures.

11.3.8 Amphipod *Ericthonius brasiliensis*

An amphipod with the first antenna measuring about half the body length, while the second antenna is slightly longer (Myers & McGrath 1984) (Figure 11.3 E). Notably, it possesses small coxae (side plates). This species differs from others in the same genus due to its comparatively smaller eyes (Milne & Griffiths 2013). *Ericthonius brasiliensis* builds muddy tubes on hydroids and other fouling species (Mead et al. 2011a). Described from the North Atlantic, it is now widely distributed in warm seas and may potentially represent a species complex. It was first reported from South Africa in 1910, and now occurs from the Olifants River to Mozambique (Mead et al. 2011a).

11.3.9 CHILEAN STONE CRAB *HOMALASPIS PLANA*

A relatively large purple crab with distinctive markings on its carapace (Figure 11.3 F). Juveniles are polychromatic, a trait that may provide protection from predation (Fernández & Castilla 2000). This crab is an important fishery species in its native range, but little is known about the habitat preference or life history of this species, and no invasion history has been documented (Fernández & Castilla 2000). A single adult male was discovered in Saldanha Bay in 2017, in the same area as *Heliaster helianthus* (Section 11.3.39), under a pier within Small Bay (Peters & Robinson 2018b). Subsequent subtidal and intertidal surveys in the surrounding rocky shore habitat revealed no other individuals and this species is not anticipated to survive along the open coast (Peters & Robinson 2018b).

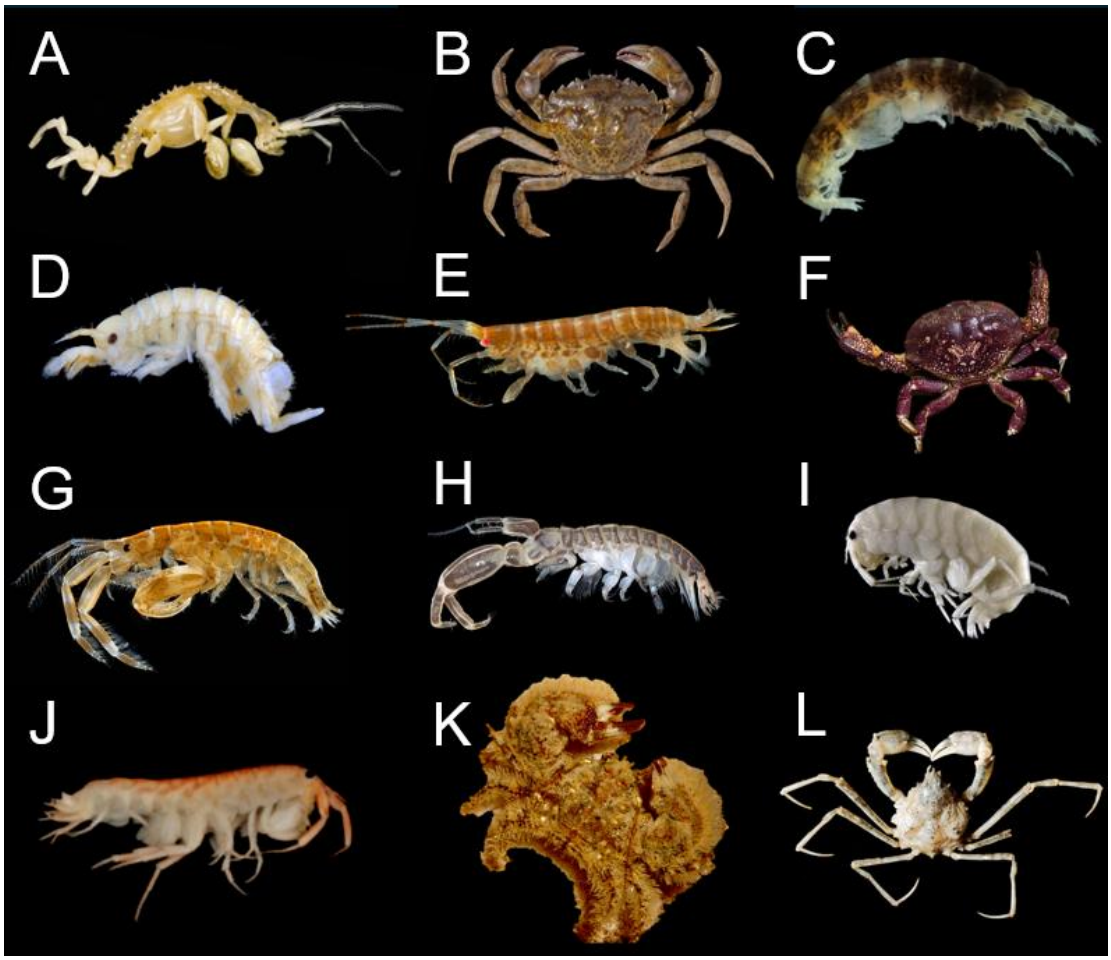


Figure 11.3. Alien arthropods recorded from Saldanha Bay and/or Langebaan Lagoon. A) Japanese skeleton shrimp *Caprella mutica* (Thierry Gosselin), B) European shore-crab *Carcinus maenas* (Nicolás Battini), C) tubular amphipod *Cerapus tubularis* (enciclovida.mx), D) wood-boring amphipod *Chelura terebrans* (Eric A. Lazo-Wasem), E) amphipod *Ericthonius brasiliensis* (Fredrik Pleijel), F) Chilean stone crab *Homalaspis plana* (K. Peters), G) amphipod *Jassa slatteryi* (plankton.image.coocan.jp), H) fat-feeler amphipod *Monocorophium acherusicum* (Eric A. Lazo-Wasem), I) shore-hopper *Orchestia gammarellus* (Hans Hillewaert), J) beach-flea *Platorchestia platensis* (Copilaș-Ciocianu et al. 2020), K) porcelain crab *Porcellana africana* (blogs.sun.ac.za), L) tuberculate pear crab *Pyromaia tuberculata* (Landschoff et al. 2022).

11.3.10 Amphipod *Jassa slatteryi*

Several closely related species occur in southern Africa, with adult males having very large second nippers that have a prominent 'thumb' projecting from the palm (Branch et al. 2022) (Figure 11.3 G). This species was first collected in South Africa in the 1950s but was incorrectly identified as the South African species *Jassa falcata*. It was only after the genus was revised that it was correctly identified as *J. slatteryi* and determined to be alien. This species is cosmopolitan, having been introduced to various locations worldwide via aquaculture and shipping (Beermann et al. 2020). Consequently, it has been termed a 'hitchhiker'.

11.3.11 Fat-feeler amphipod *Monocorophium acherusicum*

An amphipod of cylindrical body, with the last pair of legs much longer than the others (Branch et al. 2022) (Figure 11.3 H). Males have enlarged second antennae that are toothed on the lower margin. It is a globally invasive species that can tolerate a wide range of salinities and builds fragile but conspicuous u-shaped tubes where silt and detritus accumulate amongst fouling communities (Mead et al. 2011a, Albano et al. 2018). It was first recorded in South Africa from Durban Bay, based on material collected in 1915 (Barnard 1916), and was reported in Langebaan Lagoon in 2009, although the exact date of introduction is unknown (Barnard 1916, Hanekom et al. 2009).

11.3.12 Shore-hopper *Orchestia gammarellus*

This beach-flea is a green to red-brown semi-terrestrial amphipod (Fofonoff et al. 2019) (Figure 11.3 I). It occurs in the rocky shore upper intertidal, primarily in the drift-line, under rocks, debris, and vegetation (Mead et al. 2011a, Fofonoff et al. 2019). *Orchestia gammarellus* is mainly herbivorous but is also known for its scavenging behaviour. This species was first discovered in South Africa in Langebaan Lagoon during a University of Cape Town ecological survey, but was incorrectly described as a new endemic species, *Talorchestia inaequalipes* (Barnard 1951), with identification later corrected (Griffiths 1975). Alien populations have also been recorded in the Knysna Estuary and in Table Bay (Milnerton Lagoon) (Griffiths 1974, Mead et al. 2011a).

11.3.13 Beach-flea *Platorchestia platensis*

Adult males of this species are easily distinguished from those of the other introduced beach-flea *Orchestia gammarellus* due to the strongly swollen peduncle of antenna two, and the bulbous, but not flattened, distal segments on pereopod (thoracic walking leg) seven (Diemer et al. 2016) (Figure 11.3 J). This amphipod has an unknown geographic origin and confused taxonomic history but is widely distributed in tropical and temperate regions worldwide as a result of anthropogenic activities (Mead et al. 2011a, Diemer et al. 2016). It is considered an early introduction to South Africa, via solid ballast, and was first recorded in Gansbaai in 1904, but this record is dubious (Griffiths 1975, Mead et al. 2011a). Currently, it is known from Langebaan Lagoon, as well as from Cape Point eastwards to at least Algoa Bay (Diemer et al. 2016). This species is a biannual breeder, with its ability to reproduce year-round, in combination with its rapid maturity, enabling its establishment and success (Hodgson et al. 2014). It is often found in mixed colonies with native species, and potentially competitive interactions amongst the species requires further study (Diemer et al. 2016).

11.3.14 Porcelain crab *Porcellana africana*

A small porcelain crab with a flattened body and fine nodules on the left chela (Branch et al. 2022) (Figure 11.3 K). It is characterised by a dense fringe of hairs covering all limbs. Porcelain crabs, unlike true crabs, have long antennae and possess only three pairs of walking legs (with the fourth pair being small and rudimentary). Their flat bodies and pincers are adapted for dwelling in rock crevices. *Porcellana africana* was previously incorrectly identified as the European *P. platycheles* (Griffiths et al. 2018b). This species was first discovered in South Africa in relatively high numbers on Schaapen Island, Langebaan Lagoon in 2012 (Prof. George Branch *pers. comm.* 2012). However, its' date of introduction has been estimated to be between 2003 and 2009. It is now well established and abundant in Saldanha Bay, occurring across the intertidal zone under boulders and loose rocks, as well as in beds of *Mytilus galloprovincialis* mussels. In 2022, dozens of individuals were documented at the Dive School, Jetty, and Schaapen West rocky shore site, and numerous individuals were observed inhabiting kelp holdfasts at the latter site (Clark et al. 2022). Notably, in 2023, this crab was observed in significantly lower densities, only present at the Iron Ore Terminal and Jetty sites. In 2025, it was documented from the Dive School and Iron Ore Terminal sites, while no Rocky Shore survey was conducted in 2026. This species should be monitored as it has the ability to expand its range and increase in number in a relatively short time period (Griffiths et al. 2018b).

11.3.15 Tuberculate pear crab *Pyromaia tuberculata*

A spider crab with an oval carapace, featuring one anterior and three larger posterior tubercles, each covered by smaller knobs (Figure 11.3 L). Males exhibit inflated chelae. The legs are long and slender, with elongated dactyls (Landschoff et al. 2022). It was first recorded in South Africa during 2015, south of the Gouritz river mouth, at a depth of 43 m (Griffiths et al. 2018a, Landschoff et al. 2022). This crab has also been detected in Saldanha Bay by Anchor benthic macrofauna surveys from as early as 2013 but could not be identified to species level at the time using local guides (Aiden Biccard *pers. comm.* 2023). During the DNA-based survey in 2022, incidental collections of this species were undertaken, with multiple specimens sent for sequencing at the African Centre for DNA Barcoding, and identification confirmed (Payne et al. 2023a). Currently, an established population of this species occurs in Saldanha Bay.

11.3.16 Western pea crab *Rathbunixa occidentalis*

A small pinnotherid crab with a carapace width of <2.5 cm, that was first detected in Saldanha Bay (Zmarzly 1992) (Figure 11.4). It likely became established in the period between 1999 and 2004 (Clark & Griffiths 2012). The vector is unclear, although this crab was potentially introduced via ship fouling or ballast water (Clark & Griffiths 2012). This species can be free-living, although it is commonly known to live in symbiosis with other animals. Mutualistic relationships are known to facilitate the establishment and spread of introduced species (McDermott 2009). In 2026, Anchor received a request from Emma Palacios Theil (University of Łódź) for specimens of *R. occidentalis* preserved in ethanol for molecular work. Generated data will be used to compare with previous studies for the family, add barcodes for South African crustaceans, and assist with completing the phylogeny of the family. Anchor has collected these specimens and will submit them shortly.

Figure 11.4. The alien invasive Western pea crab *Rathbunixa occidentalis* (Smithsonian_marinegeo).

Continuous monitoring of the pea crab as part of the SOB survey in Saldanha Bay has been ongoing since 2004, comprising multiple monitoring stations in Big Bay, Small Bay, Langebaan Lagoon and Danger Bay. Abundance fluctuates annually, with this species being more abundant in Big Bay, followed by Small Bay (Figure 11.5). *Rathbunixa occidentalis* has been documented in Langebaan Lagoon only twice during the SOB monitoring. Notably, this crab was also found in Danger Bay in 2015, suggesting the potential for the species to extend its distribution beyond Saldanha Bay. However, subsequent sampling has not been conducted in this region to verify the ongoing presence of this crab. Despite continued sampling efforts in each survey, certain sites in the Bay consistently show no presence of crabs. This could indicate that the crab has not yet extended to these sites, or that it may prefer a specific habitat type. The data suggest that this pea crab prefers deeper-water habitats and is generally scarce, or occurs in lower densities, near the Iron Ore and Multi-Purpose Terminals. While this species was not detected at any sampling stations during the 2023 survey (Clark et al. 2023), the highest number of individuals recorded since 2004 was in 2026, with 1 520 documented in Big Bay, and 816 in Small Bay (Figure 11.5). The impact and role of the pea crab in Saldanha Bay's benthic community remains undetermined, highlighting the necessity for more in-depth studies and potential management actions.

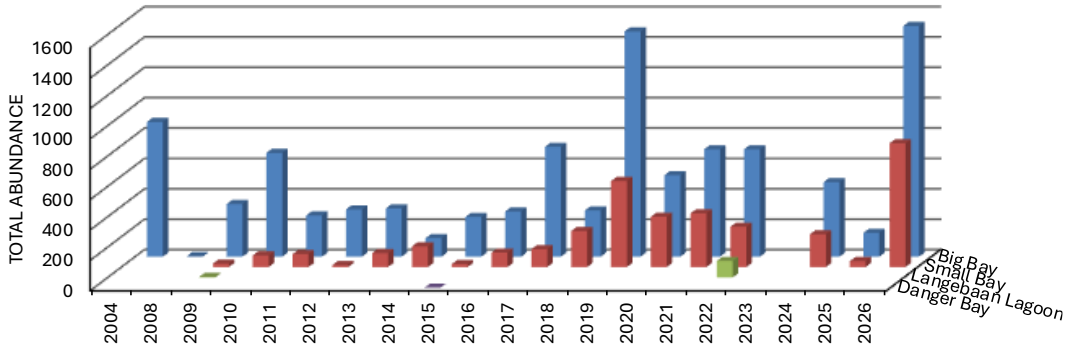


Figure 11.5. Total abundance of the Western pea crab *Rathbunixa occidentalis* in Big Bay, Small Bay, Langebaan Lagoon and Danger Bay from the period 2004–2026. Note that Langebaan Lagoon and Danger Bay were first sampled in 2004 and 2014, respectively. No data were collected in the period 2005–2007. 'ND' denotes that no data were collected in that region for that year.

11.3.17 Striped barnacle *Amphibalanus venustus*

A barnacle often found under low-shore boulders in South Africa that has thin pink-red stripes of uniform width, and reaches 5–8 mm (Biccard 2012, Branch et al. 2022) (Figure 11.6 A). Originating from the western North Atlantic, this species was first recorded from South Africa in 1938 from Durban Harbour (Mead et al. 2011a). Likely introduced via ship fouling, *A. venustus* now occurs from Saldanha Bay to Mozambique (Mead et al. 2011a, Biccard 2012, Matikinca & Robinson 2024). In South Africa, its depth range is low intertidal to 66 m (Biccard 2012). A recent study found that this species may decline due to possible future anthropogenic cooling and acidification (Matikinca & Robinson 2024).

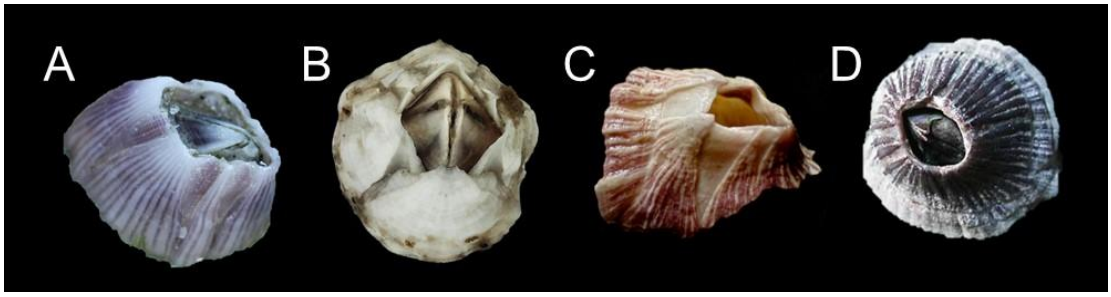


Figure 11.6. Alien barnacles recorded from Saldanha Bay and/or Langebaan Lagoon. A) striped barnacle *Amphibalanus venustus* (Even Dankowicz), B) Pacific barnacle *Balanus glandula* (Cédric d'Udekem d'Acoz & Francis Kerckhof), C) pitted/triangle barnacle *Balanus trigonus* (litoraldegranada.ugr.es), D) perforated barnacle *Perforatus perforatus* (Peter Barfield).

11.3.18 Pacific barnacle *Balanus glandula*

A conical barnacle, off-white to yellowish in colour, with a calcareous base (Branch et al. 2022) (Figure 11.6 B) with six vertically ridged shell plates and no stripes. The junction between the four upper shell plates forms an M-shaped line. It has been in South Africa since at least the early 1990s, but was first recognised in 2008 (Laird & Griffiths 2008, Simon-Blecher et al. 2008). This abundant species ranges from past Cape Point to Elands Bay on the west coast (Laird & Griffiths 2008, Robinson et al. 2015a), displacing populations of the indigenous toothed barnacle *Chthamalus dentatus* (Simon-Blecher et al. 2008).

11.3.19 Pitted/triangle barnacle *Balanus trigonus*

A mottled pink-purple barnacle with six ribbed shell plates and a triangular operculum (Biccard 2012) (Figure 11.6 C), *Balanus trigonus* is well established along the entire South African coastline (Biccard 2012). First recorded from Knysna Estuary in 1947, this species is thought to have been introduced to South Africa via shipping activity (Millard 1949). This barnacle is a common fouling species, and is prolific in warmer waters (e.g., in harbours and estuaries), where it is also able to tolerate reduced salinity levels (Millard 1949, Biccard 2012).

11.3.20 Perforated barnacle *Perforatus perforatus*

A volcano-shaped barnacle with a small oval aperture and calcareous base (Figure 11.6 D), the operculum is deeply immersed within the aperture and terga with short, pointed beak (Skewes 2008). First reported from South Africa on rocks in Simonstown (Gruvel

1907), the presence of this species in the country was confirmed after attaining fresh material from Mossel Bay (Biccard 2012). This barnacle was first detected in Saldanha Bay during 2011, but was incorrectly identified as *Menesiniella regalis* in Biccard & Griffiths (2016) (Aiden Biccard *pers. comm.* 2023). Subsequently, it has frequently been encountered on the rocky shore during SOB surveys, although in relatively low numbers.

11.3.21 Disc lamp shell *Discinisca tenuis*

A disc-shaped brachiopod with a semi-transparent, hairy, fringed shell (Figure 11.7). It was first recorded in 2008 on oysters grown in Saldanha Bay, and later reported on Schaapen Island (Haupt et al. 2010, Peters et al. 2014). This species is endemic to Namibia and is thought to have been introduced to South Africa via cultured oyster imports (Haupt et al. 2010). It can reportedly reach very high densities and could become a significant fouling species in Saldanha Bay.

Figure 11.7. The alien invasive disc lamp shell *Discinisca tenuis* (Prof. C.L. Griffiths).

11.3.22 Fouling moss animal *Bugula neritina*

A bryozoan that forms reddish-purple tuft-like colonies, and is a common fouling species with a local range from Port Nolloth to Durban (Boonzaaier 2017) (Figure 11.8 A). Globally, it has a cosmopolitan distribution in warm-temperate and tropical regions (Boonzaaier 2017). Of unknown origin, it was first documented in South Africa in 1944.

11.3.23 Fan-shaped moss animal *Bugulina flabellata*

An upright, bushy, branched bryozoan about 2–5 cm in height, that is beige in colour (Ballerstedt 2004) (Figure 11.8 B). Globally, it has a widespread distribution in both warm and cold temperate waters in both hemispheres (Gordon & Mawatari 1992). It is a well-known fouling species that is commonly attached to ship hulls and other bryozoans (Florence et al. 2007, Boonzaaier 2017). One of the earliest records of introduced species in South Africa (Mead et al. 2011b), being collected in 1880, it is known to occur from Port Nolloth to Plettenberg Bay (Florence et al. 2007).

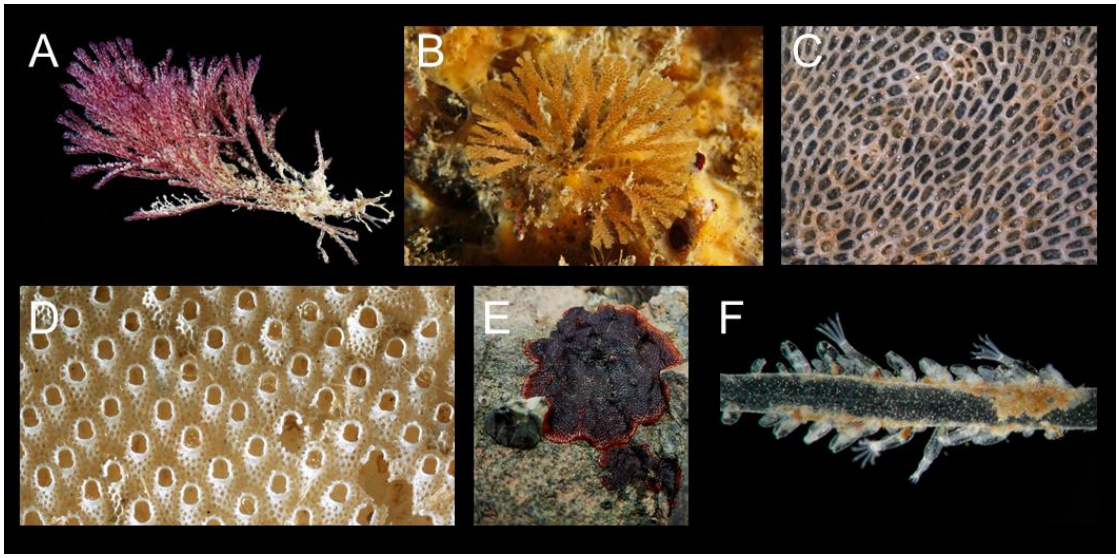


Figure 11.8. Alien invasive bryozoans recorded from Saldanha Bay and/or Langebaan Lagoon. A) fouling moss animal *Bugula neritina* (Dean Janiak), B) fan-shaped moss animal *Bugulina flabellata* (bie.ala.org.au), C) *Conopeum seurati* (Alison M. Fortunato), D) red crust *Cryptosula pallasiana* (Eric A. Lazo-Wasem), E) red-rust bryozoan *Watersipora subtorquata* (Anchor & wikimedia.org), F) spaghetti bryozoan *Amathia verticillata* (noticias.cebimar.usp.br).

11.3.24 Bryozoan *Conopeum seurati*

A bryozoan forming brownish-white sheets, irregular incrustations or spherical balls of zooids depending on the substrate upon which it settles (britishbryozoans.myspecies.info) (Figure 11.8 C). Native to brackish water, lagoons and estuaries in Europe, this species was first recorded in South Africa in Saldanha Bay in 2001, although it has probably been present for decades, if not centuries, before then (Poluzzi & Sabelli 1985, Awad et al. 2005). This bryozoan potentially also occurs in Zandvlei Lagoon (False Bay), although proper identification is required (Mead et al. 2011a).

11.3.25 Red crust *Cryptosula pallasiana*

A bryozoan that forms pink, white or orange encrusting colonies and has been reported from numerous harbours around the globe (Occhipinti Ambrogi & D'Hondt 1981, Gordon & Mawatari 1992) (Figure 11.8 D). This species was first recorded in South Africa from Table Bay harbour, based on specimens collected during 1947–1949 (Millard 1951). It was later recorded in Simonstown and Saldanha Bay, however, it is most likely widespread throughout South African estuaries (Henschel et al. 1990, Awad et al. 2005, Mead et al. 2011a).

11.3.26 Red-rust bryozoan *Watersipora subtorquata*

A common fouling bryozoan of unknown native range which forms crusty red-rust patches on rocks from the intertidal down to 45 m in South Africa (Florence et al. 2007, Mead et al. 2011a, Boonzaaier 2017) (Figure 11.8 E). It was first reported in the country in 1937, with a current known distribution range from Langebaan Lagoon to Algoa Bay as of 2017 (Mead et al. 2011b, Boonzaaier 2017). This colony-forming species favours

environments of low physical stress and is tolerant to antifouling biocides used on vessel hulls (Floerl & Inglis 2005, Malherbe 2011).

11.3.27 Spaghetti bryozoan *Amathia verticillata*

This erect, soft-bodied bryozoan has a cosmopolitan distribution and unknown origin (Figure 11.8 F). It is a generalist species that forms extensive and massive colonies which can spread to adjacent areas via larvae or colony detachment. This species exhibits an interesting deciduous pattern, with abundance peaking in summer and diminishing in late winter (Guerra-García et al. 2024). It was first recorded in South Africa from Langebaan Lagoon in 2023 (Ackland et al. 2025a), and was later documented in the Port of Ngqura by Anchor (Bovim et al. 2024). This species was likely introduced via biofouling, and a seasonal monitoring plan has been recommended due to its current extensive invasion in Langebaan Lagoon (2 333 colonies recorded, some more than 2 m long), and potential environmental and socio-economic impacts (Ackland et al. 2025a). Of specific concern is the presence of this species within meadows of the regionally endangered seagrass *Zostera capensis*, and its potential shading impact.

11.3.28 Dirty sea squirt *Ascidiella aspersa*

A solitary ascidian that is semi-transparent, with a body wall that is normally white with pale red siphons (Mead et al. 2011a) (Figure 11.9 A). It is commonly found in dense, unfused aggregations with attached detritus (Curtis 2005). This species has been recorded from Saldanha Bay, Cape Town, Hout Bay and Gqeberha (Monniot et al. 2001), and seems to be restricted to harbour environments (Monniot et al. 2001, Robinson et al. 2004b).

11.3.29 Ascidian *Asterocarpa humilis*

A large and conspicuous solitary ascidian, with a leathery tunic often covered in various epibionts (Monniot et al. 2001) (Figure 11.9 B). In South Africa, this species was first recorded from ropes in Hout Bay (Monniot et al. 2001) and has been recorded from Saldanha Bay, False Bay and Gqeberha (Loureiro et al. 2021).

11.3.30 Golden star ascidian *Botryllus schlosseri*

A colonial ascidian that is variable in colour, but often dark brown to black with zooids arranged in separated, star-shaped clusters (Branch et al. 2022) (Figure 11.9 C). The colonies are small, thin and encrusting (Monniot et al. 2001). This species was first recorded from South Africa, based on specimens collected in 1946 from Durban harbour (Millar 1955). It is currently found in many harbours, from Alexander Bay to Gqeberha, including in Saldanha Bay (Monniot et al. 2001, Loureiro et al. 2021). This ascidian often colonizes other species, and may possibly have a negative impact on indigenous kelp and the eelgrass *Zostera capensis* (Griffiths et al. 2009).

11.3.31 Sea vase *Ciona robusta*

This species is a tall and cylindrical solitary ascidian with a soft, transparent test (Figure 11.9 D). It was initially misidentified as *Ciona intestinalis*, which was found to comprise two morphologically separate species. Originally introduced from the North Atlantic prior

to 1955, this species now occurs in harbours along the entire South African coastline (Mead et al. 2011a). This tunicate is an economically important pest and is known to smother mussels on aquaculture infrastructure, especially mussel ropes (Robinson et al. 2005a).

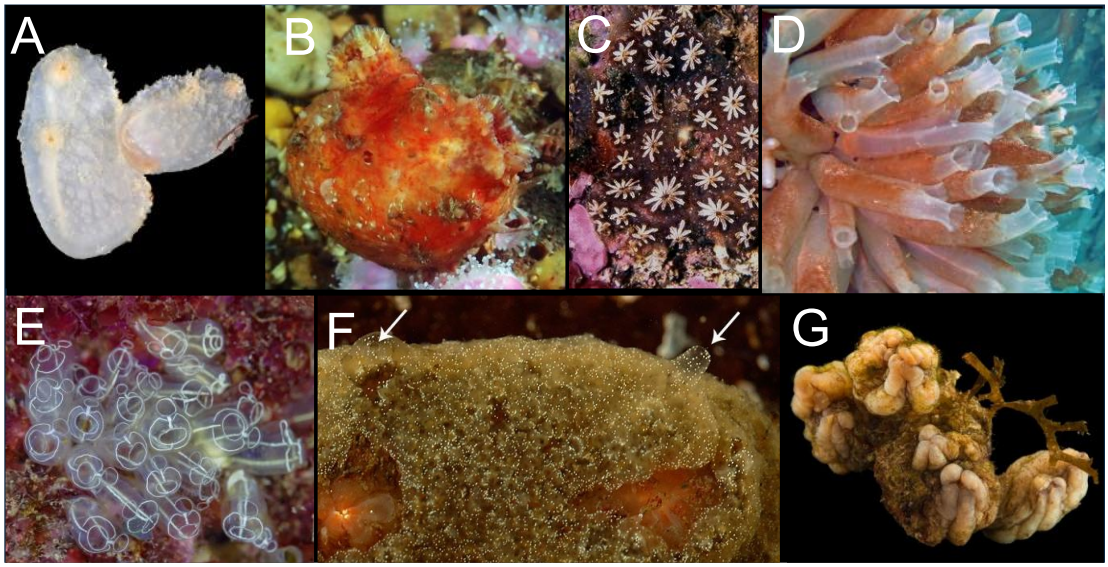


Figure 11.9. Alien invasive ascidians recorded from Saldanha Bay and/or Langebaan Lagoon. A) dirty sea squirt *Ascidiella aspersa* (Paul Sterry/Nature Photographers), B) *Asterocarpa humilis* (B. Guichard/PatriNat), C) golden star ascidian *Botryllus schlosseri* (Massimiliano DE MARTINO), D) sea vase *Ciona robusta* (Anchor), E) bell ascidian *Clavelina lepadiformis* (Diego Delso), F) gossamer ascidian *Diplosoma listerianum* (Prof. C.L. Griffiths), G) pleated sea squirt *Styela plicata* (gulfspecimen.org).

11.3.32 Bell ascidian *Clavelina lepadiformis*

A colonial ascidian that grows up to 2 cm high, with transparent zooids that have a white ring around the pharynx and have pale yellow or white longitudinal lines along the endostyle and dorsal lamina (Riley 2008) (Figure 11.9 E). This species is often found attached to the underside and sides of boats and jetties (Mead et al. 2011a). It was first recorded in South Africa from the Knysna Estuary and Port Elizabeth Harbour (Monniot et al. 2001), and was subsequently documented from other locations along the South African coastline, including Saldanha Bay (Rius et al. 2014).

11.3.33 Gossamer ascidian *Diplosoma listerianum*

A yellow to dark grey colonial ascidian that forms thin, fragile, jelly-like sheets (Branch et al. 2022) (Figure 11.9 F). It is commonly mistaken for a sponge or completely overlooked. This species is believed to have been introduced to South Africa from Europe prior to 1949, and is found encrusting on all types of substrata on sheltered shores between Alexander Bay and Durban (Monniot et al. 2001, Picker & Griffiths 2011).

11.3.34 Pleated sea squirt *Styela plicata*

This solitary ascidian has a characteristic thick, tough tunic (Mead et al. 2011a) (Figure 11.9 G). It is a common species in marinas and harbours of warm and temperate oceans,

and is usually found at high densities (Pineda et al. 2011). First documented from South Africa in Durban in 1951, this species is now very abundant at several locations along the coast (Mead et al. 2011a).

11.3.35 European daisy anemone *Cereus pedunculatus*

A small anemone, measuring 2.5–30 mm in pedal disc diameter, featuring an off-white base and a green-brown upper portion covered with adhesive discs for the attachment of gravel or shells (Figure 11.10 A). The oral disc is a greenish colour, but paler around the mouth. When disturbed, it has the ability to eject sticky threads (Branch et al. 2022). In South Africa, it is only known from Langebaan Lagoon (first recorded in 2001), where it occurs intertidally in seagrass beds (*Zostera capensis*), attached to rocks covered by sand, and in loose rocks resting on fossilized oyster beds (Acuña et al. 2004, Robinson et al. 2004b, Picker & Griffiths 2011, Robinson & Swart 2015). Originally identified as the alien rooted anemone *Sagartia* (now *Cylista*) *ornata*, South African material has recently been correctly identified as *Cereus pedunculatus*, with likely vectors being ballast water or mariculture (Andersen et al. 2026). This updated record marks the first documented occurrence of this species outside its native range (European North Atlantic and Mediterranean Sea) (Andersen et al. 2026). In South Africa, this anemone has been found to alter community structure, supporting a higher invertebrate abundance, biomass and diversity where it occurs (Robinson & Swart 2015).

11.3.36 Hydroid *Coryne eximia*

A hydroid characterised by its red or pink colour, growing up to 15 cm, although typically reaching about half that size (Figure 11.10 B). The branches emerge from one side of each stem, and are mostly smooth with small, irregular ringed sections. At the end of each branch, there is a polyp featuring a cluster of knobbed tentacles (Barrett & Yonge 1958, Schuchert 2023). This cosmopolitan fouling species commonly occurs in shallow water down to a depth of 25 m on the anchoring chains of buoys, rafts, mussels, rocks and seaweed (Millard 1975, Schuchert 2005). It was first discovered in South Africa in 1946, and is found along the west coast, including in Langebaan Lagoon.

11.3.37 Pinkmouth hydroid *Ectopleura crocea*

This hydroid has a long, unbranched stem that is encased in a sheath (Branch et al. 2022) (Figure 11.10 C). It has a single large pink and white polyp with short tentacles forming a ring around the mouth. A second ring of larger tentacles is also present that has grape-like bunches of reproductive sporosacs. *Ectopleura crocea* is a North Atlantic species with a complex taxonomic history. It was first recorded in South Africa from Durban Harbour in 1947 and was inadvertently re-described as a new species (*Tubularia warreni*) (Mead et al. 2011a). This hydroid is common in dock areas on pylons and ships hulls (Millard 1975).

11.3.38 Thin-walled obelia *Obelia dichotoma*

A light brown colonial hydroid that appears solitary, or forms fan to bush-like colonies (Figure 11.10 D). The stems and branches have a slight zig-zag appearance, and nodes are marked with three to four annulations (Richards 2007). It is common in dock areas and frequently found on other hydroids and algae (Millard 1975). The first collection of

this species from South Africa likely occurred in 1938 (Mead et al. 2011a). It is found along the entire South African coastline, including in natural habitats (Robinson et al. 2020).

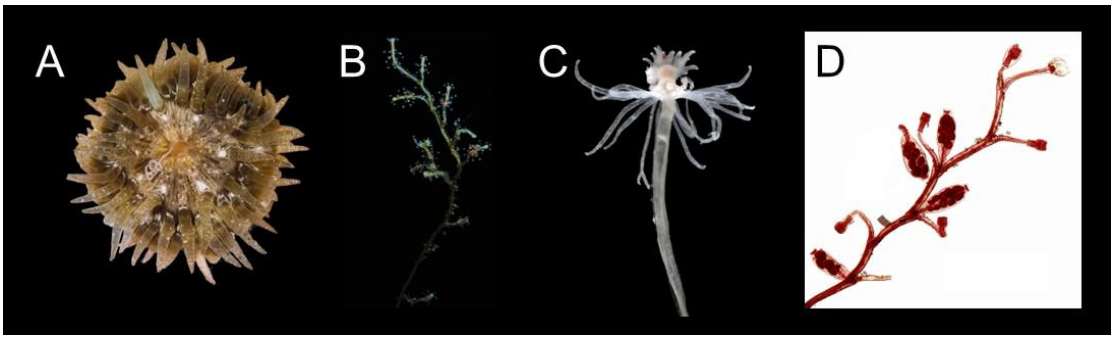


Figure 11.10. Alien and naturalised cnidarians recorded from Saldanha Bay and/or Langebaan Lagoon. A) European daisy anemone *Cereus pedunculatus* (Andersen et al. 2026), B) hydroid *Coryne eximia* (Peter Schuchert), C) pinkmouth hydroid *Ectopleura crocea* (Eric A. Lazo-Wasem), D) thin-walled obelia *Obelia dichotoma* (Daniel J. Drew).

11.3.39 South American sunstar *Heliaster helianthus*

A large sea star that can grow up to 20 cm in diameter, with up to 40 arms (Madsen 1956, Barahona & Navarrete 2010) (Figure 11.11). In its native range, it is a keystone species, playing an important role in structuring intertidal and subtidal communities (Paine et al. 1985, Navarrete & Manzur 2008). This species has planktotrophic larvae with extended longevity, which allows for long distance dispersal, and is a trait that could facilitate invasion (Navarrete & Manzur 2008, Castilla et al. 2013). A single large adult was discovered in Saldanha Bay in 2015, under a pier within Small Bay, close to Hoedjiesbaai (Peters & Robinson 2018b). Subsequent subtidal and intertidal surveys in the surrounding rocky shore habitat revealed no other individuals (Peters & Robinson 2018b). To date, no other individuals have been detected in the Bay.

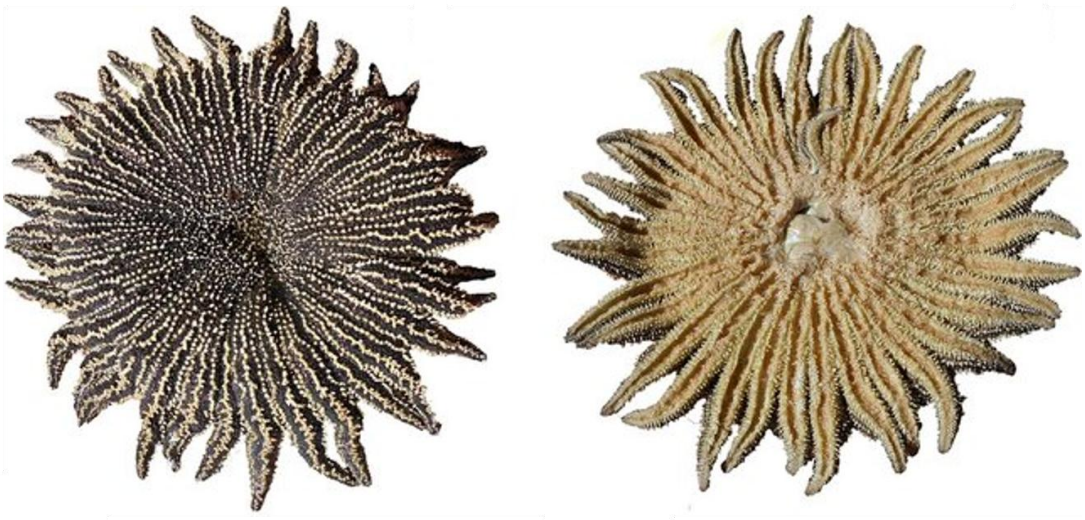


Figure 11.11. The alien South American sunstar *Heliaster helianthus* (Peters & Robinson 2018b).

11.3.40 Pacific oyster *Magallana gigas*

An oyster of length twice the breadth, measuring around 200 mm with strong concentric corrugations and wave-like scallops (Figure 11.12 A). Internally white with no lustre, and one valve slightly indented below apex (Branch et al. 2022). This oyster was introduced to the Knysna Estuary in South Africa, in the 1950s, with the intention to farm. This species has been farmed in the Kowie and Swartkops estuaries, as well as at three marine locations including Algoa Bay, Saldanha Bay and Alexander Bay (Robinson et al. 2005b). Initially, this species was not considered an invasive threat as the oyster seemed unable to settle and reproduce successfully in the local environmental conditions. However, farmed populations have spread throughout the country. Using DNA sequencing, Robinson et al. (2005b) confirmed the presence of three naturalised populations of *M. gigas* in South Africa (specifically the Breede, Knysna and Goukou estuaries). In Saldanha Bay, *M. gigas* was originally farmed in the Seafarm Dam east of the Iron Ore Terminal 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.

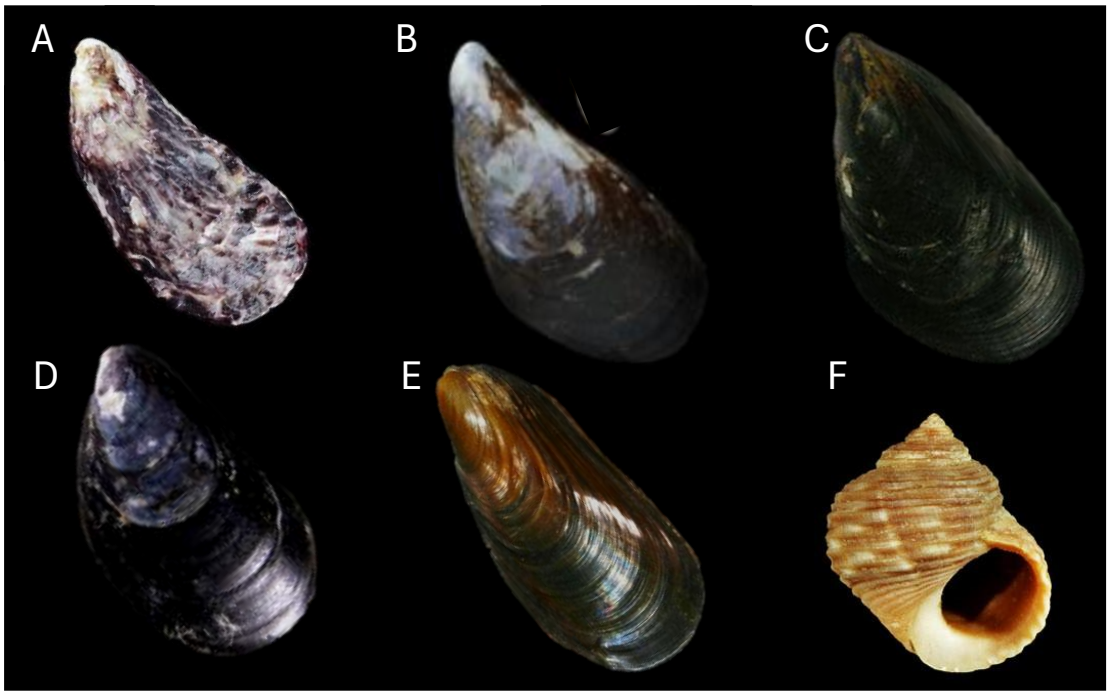


Figure 11.12. Alien invasive molluscs recorded from Saldanha Bay and/or Langebaan Lagoon. A) Pacific oyster *Magallana gigas* (mollusca.co.nz), B) blue mussel *Mytilus edulis* (Rudolf Kapeller), C) Mediterranean mussel *Mytilus galloprovincialis* (malacopics.nl), D) bay mussel *Mytilus trossulus* (Amgueddfa Cymru–National Museum Wales), E) bisexual mussel *Semimytilus patagonicus* (Joop Trausel & Frans Slieker), F) British periwinkle *Littorina saxatilis* (Joop Trausel & Frans Slieker).

11.3.41 Mytilid mussel species complex

Prior to the 2022 DNA-based survey, the Mediterranean mussel *Mytilus galloprovincialis* and the bisexual mussel *Semimytilus patagonicus* were believed to be the only alien mussels present in Saldanha Bay (Payne et al. 2023a). However, metabarcoding detected three *Mytilus* species (*M. galloprovincialis*, *M. edulis* and *M. trossulus*) from wall

scrape samples. These are collectively referred to as the mytilid species complex, which is subject to extensive research worldwide (Ab Rahim et al. 2016), but had not been previously documented in South Africa. While these species have largely distinct distributions, all of them are globally widespread and they can hybridize where their ranges overlap, which makes identification difficult (Ab Rahim et al. 2016). In addition, the presence of *M. edulis* and *M. trossulus* may have been previously obscured by the dominance of *M. galloprovincialis*. Although the detection of this complex is concerning, further morphological taxonomic and DNA investigation is required to confirm whether the presence of *M. edulis* and *M. trossulus* in South Africa is accurate.

BLUE MUSSEL MYTILUS EDULIS

Usually purple, blue or dark brown, this mussel has a smooth inequilateral shell marked by concentric growth lines originating from the hinge, and a pearl-white interior (Bonham & Roberts 2022) (Figure 11.12 B). It can occur subtidally and intertidally on rocky shores and has a high tolerance for elevated sediment levels. Consequently, this mussel may assist with removing sediment from the water column (FAO 2023). The detection of this species through the DNA-based survey in 2022 marked the first genetic record of this species in South Africa and Saldanha Bay. In 2026, Anchor collected further mytilid specimens and sent them to Pavel Nekhoroshkov (Joint Institute for Nuclear Research, Russia) for genetic testing. Preliminary results confirm the presence of *Mytilus edulis*, or a hybrid of this species and *M. galloprovincialis*, in Saldanha Bay.

MEDITERRANEAN MUSSEL MYTILUS GALLOPROVINCIALIS

A smooth broad mussel, typically black or blue (Figure 11.12 C), it is easily confused with the indigenous black mussel *Choromytilus meridionalis*, but differentiated by the presence of pits in the resilial ridge (Branch et al. 2022). First detected from South Africa in 1979, its presence was only confirmed in 1984 (Mead et al. 2011a). At this point, 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. 2005a). This mussel has caused significant ecological impacts, including the displacement of indigenous species and an increase in intertidal mussel biomass. Positive changes include the increased available habitat for infauna, and food supply for intertidal predators (Robinson et al. 2005a). It is commercially cultured in Saldanha Bay, and elsewhere, and is widely exploited by recreational and subsistence fishers (Robinson et al. 2005a).

BAY MUSSEL MYTILUS TROSSULUS

An elongated mussel with a smooth blue-black surface with concentric lines and a glossy brown-black coating (Coe 1945, Kozloff 1974) (Figure 11.12 D). The detection of this species through the DNA-based survey in 2022 marked the first genetic record of this species in South Africa and Saldanha Bay (Payne et al. 2023a). However, it is recommended that further specimens are examined morphologically and sent for DNA barcoding.

BISEXUAL MUSSEL SEMIMYTILUS PATAGONICUS

A small elongate species that is hermaphroditic and brown or blackish-green in colour (Branch et al. 2022) (Figure 11.12 E). This reef-forming mussel was reported from South Africa in 2010, and it has become a dominant intertidal organism on wave-exposed lower shores, with a present geographic range from Groenriviersmond in the north to Bloubergstrand in the south (Mead et al. 2011a, de Greef et al. 2013) This species likely migrated southward from Namibia, either through shipping as a new invasion, or via range expansion from the Namibian population (de Greef et al. 2013).

11.3.42 BRITISH PERIWINKLE *LITTORINA SAXATILIS*

A globular periwinkle, of 5–8 mm in size, with a round aperture that flanges forward at the front (Figure 11.12 F). Colour is variable, ranging from uniformly off-white to pale brown with darker rings (Branch et al. 2022). *Littorina saxatilis* was first recorded in South Africa in 1974 (Day 1974), and the only known populations are those in the Langebaan and Knysna lagoons (Hughes 1979, Robinson et al. 2004b, Picker & Griffiths 2011). In its home range, this species occupies crevices on rocky shores (Gibson et al. 2001). However, 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). This species only occurs in the upper reaches of Langebaan Lagoon, between Bottelary and Churchhaven, and has not spread further in the last 20 years (Robinson et al. 2004b). It is not considered to be a major threat to the Lagoon or Bay ecosystems.

11.3.43 FRAGILE UPRIGHT CODIUM *CODIUM FRAGILE*

An intertidal green algae with distinctive thick, cylindrical spongy branches (Branch et al. 2022) (Figure 11.13 A). The first recorded date of introduction and its presence in South Africa is undetermined, primarily because it can be mistaken for the native species found from Namibia's west coast to Plettenberg Bay on the east coast (Stegenga et al. 1997). Additionally, there has been uncertainty surrounding its nomenclature. This species thrives in various habitats, including rocky and sandy environments, where it attaches to natural and artificial solid surfaces (Bulleri & Airoldi 2005, Geraldi et al. 2014). *Codium fragile* is also known to invade kelp beds and *Zostera marina* eelgrass beds (Ramus 1971, Scheibling & Gagnon 2006). It is considered euryhaline (tolerant of fluctuating salinities) and tolerant of a wide range of temperatures (-2 to 30°C) and desiccation, all traits that could contribute towards its introduction and invasion success (Malinowski & Ramus 1973, Hanisak 1979, Schaffelke & Deane 2005). While the specific impacts in Saldanha Bay are yet to be quantified, it's worth noting that in other invaded locations this species has both positive and negative effects, serving as an important food source and habitat for numerous invertebrates (Scheibling & Anthony 2001, Cruz-Rivera & Hay 2001, Harris & Jones 2005). Comprehensive surveys and genetic analysis are essential to determine the precise identity and distribution of this species in Saldanha Bay and Langebaan Lagoon.

11.3.44 RED ALGAE *ANTITHAMNIONELLA SPIROGRAPHIDIS*

A small red algal species with features only visible under a microscope (Figure 11.13 B). It has been documented from the south coast in the Kowie estuary, and in sheltered areas of Saldanha Bay attached to jetties in 1989 (Stegenga et al. 1997). It is commonly

associated with harbours and docks, with its success as an invader attributed to its vegetative, rapid reproduction (Wollaston 1968).

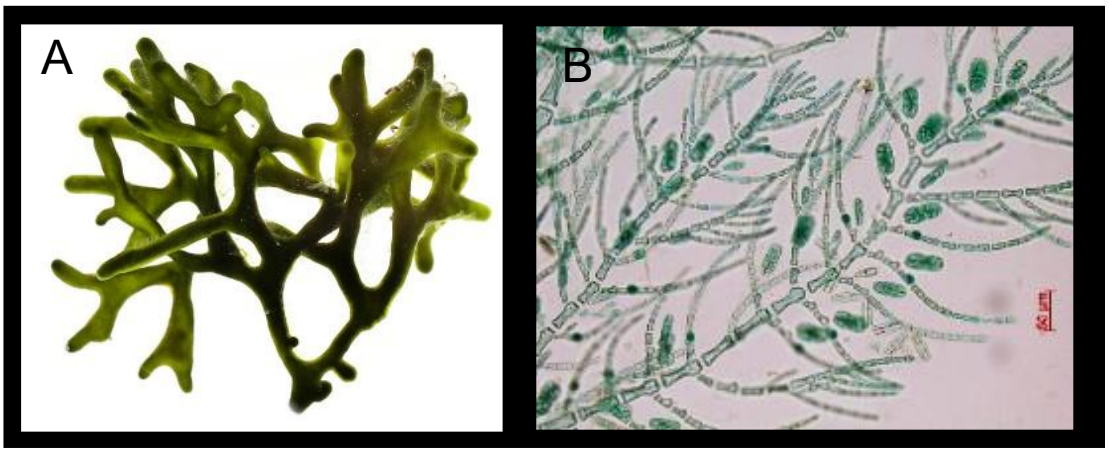


Figure 11.13. Alien invasive marine plants recorded from Saldanha Bay and/or Langebaan Lagoon. A) fragile upright codium *Codium fragile* (coralaxy.de), B) red algae *Antithamnionella spirographidis* (southafrseaweeds.uct.ac.za).

12 AQUATIC MACROPHYTES

12.1 INTRODUCTION

Macrophytes (large aquatic plants that grow in or near water) fulfil important ecological functions in estuarine habitats. The area coverage and species composition of macrophyte habitats are used to assess the condition of South African estuaries and inform management and conservation priorities. The Langebaan estuarine lagoon has the highest macrophyte species richness across all South African estuaries (Adams et al. 2026b) and contains four distinct intertidal habitats: seagrass beds, such as those of the eelgrass *Zostera capensis*; salt marsh, dominated by cordgrass *Spartina maritima* and perennial glasswort *Sarcocornia perennis*; reeds and rushes, such as common reed *Phragmites australis*, bulrush *Typha capensis*, and the dune slack rush *Juncus kraussi*; and lastly sandflats, which are largely unvegetated and dominated by the sand prawn *Kraussillichirus kraussi* and the mudprawn *Upogebia capensis* (Siebert & Branch 2005). Each of these habitats and their associated vegetation are important, and their interconnectivity facilitates essential ecological dynamics. For more information on the intrinsic ecological and economic value of macrophytes, see Clark et al. (2025).

12.1.1 SEAGRASSES

Seagrasses, belonging to the Monocotyledon clade, are marine plants that flower and seed under water, forming vast meadows (Short et al. 2011). Seagrasses are critical ecosystem engineers in marine and estuarine environments across the world and in South Africa. They provide habitat, shelter and nursery and foraging areas for a number of commercially important fish and invertebrate species (Adams 2016). They also provide valuable ecosystem services such as nutrient cycling, coastal resilience and carbon sequestration (Costanza et al. 1997, 2014). Along with mangroves and salt marshes, seagrasses are recognised as blue carbon ecosystems due to their exceptional capacity to sequester and store carbon.

The dominant seagrass in South Africa is the Cape Dwarf eelgrass *Zostera capensis*, which occurs in 62 sheltered estuaries distributed along the approximate 3 000 km national coastline (Adams 2016) (Figure 12.1). Protected meristems, a strong root system, and flexible leaves enable *Z. capensis* to thrive under the strong tidal conditions and periods of emersion that are characteristic of intertidal zones (Adams 2016). *Z. capensis* is found predominantly in permanently open estuaries that exhibit moderately brackish to marine conditions. Langebaan Lagoon supports one of the largest known populations of *Z. capensis* and is one of only two formally protected estuaries in South Africa where the species occurs. However, the conservation and management of *Z. capensis* is challenging because the extent and condition of its populations, which occur as discrete patches or stands that collectively form larger meadows, exhibit a high degree of variability in extent over time. Seagrass cover and biomass fluctuate in response to natural drivers, including tidal exchange and flood events, as well as anthropogenic pressures such as eutrophication, bait digging, boating activity, and trampling (Adams et al. 2026b). Consequently, distinguishing natural variability from human-induced change can be difficult.

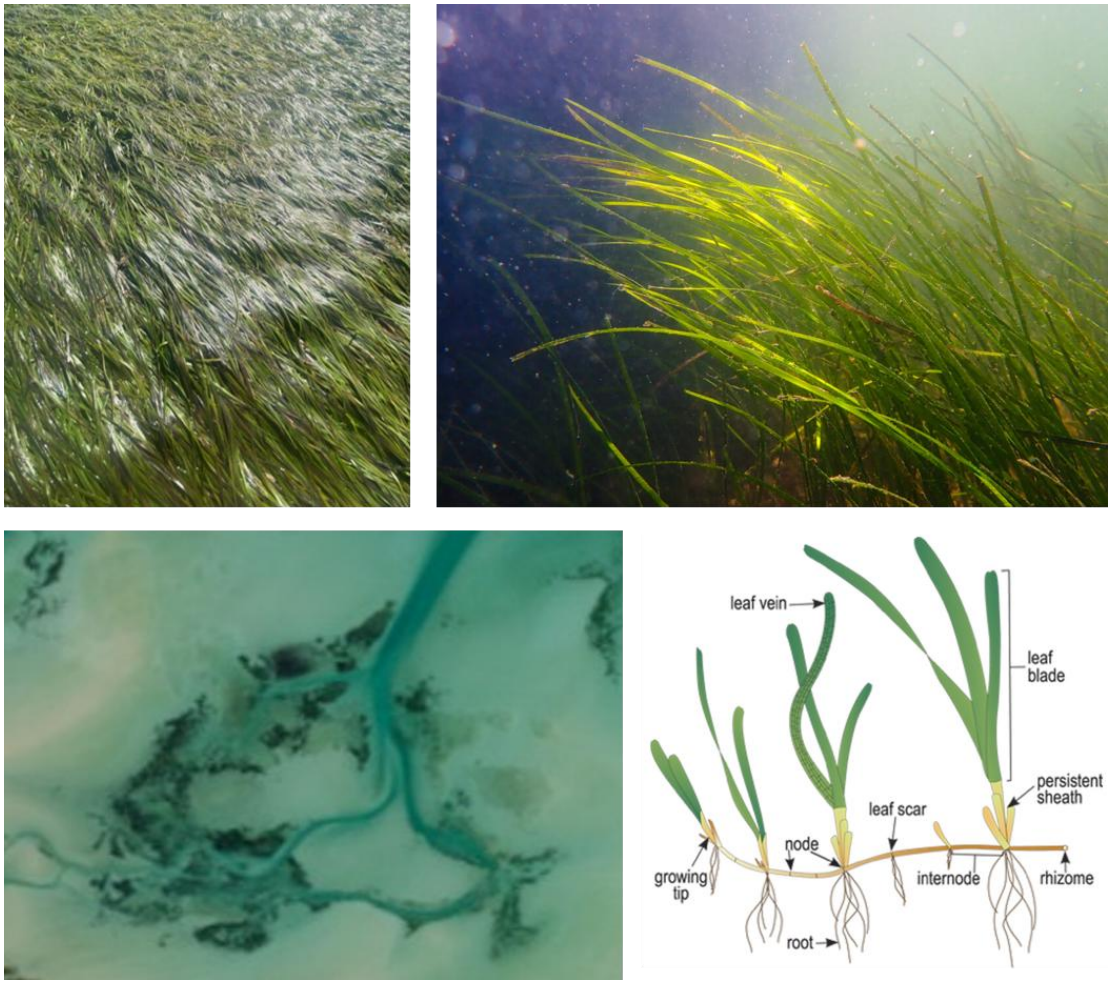


Figure 12.1. Anatomy and visual characteristics of *Zostera capensis* (Photo credit: Dr Katie Watson; "European Space Agency (ESA), Copernicus Sentinel-2 Imagery (2026)).

Among macrophyte communities, seagrasses are particularly vulnerable to these pressures due to their dependence on clear, nutrient-balanced coastal waters. Increased pollution and nutrient input to coastal systems from stormwater runoff creates and intensifies eutrophic conditions (Lawrence 2025). During these conditions, light and oxygen availability are reduced while sulphide levels in sediments increase, all of which have detrimental effects on the photosynthetic rates, metabolism and growth of seagrasses (Lawrence 2025). Anthropogenic-induced climate change also represents an increasingly important threat to seagrasses, with predicted increases in the frequency, duration and intensity of extreme climate events, such as marine heat waves. Marine heat waves have been shown to induce acute physiological stress and subsequent mass seagrass mortality (Smale et al. 2019) and otherwise exacerbate local stressors (Kendrick et al. 2019), potentially pushing already vulnerable seagrass populations past their functional thresholds. Worst case-scenarios might include contractions, range shifts or even local extinctions (Hyndes et al. 2017, Nguyen et al. 2021).

At a national scale, most seagrasses meadows remain under significant pressure, while accelerating rates of meadow loss and degradation are occurring globally (Waycott et al. 2009). As of 2010, *Z. capensis* is listed as Vulnerable on the global IUCN Red List and Endangered in South Africa in the Red Data List of Species (Short et al. 2010, 2011, Adams & van der Colff 2018), with seagrass cover around South Africa having declined by

approximately 33% from the 1930s to 2020 (Raw et al. 2023). Loss of *Z. capensis* cover in South Africa, and more specifically in Langebaan Lagoon, is largely anthropogenically driven from activities such as dredging, development, changes in flow dynamics and eutrophication (Adams 2016).

12.1.2 SALT MARSH

Salt marshes are intertidal coastal wetlands that are regularly flooded and drained by saline water through tidal cycles. They typically occur on deep mud and peat substrates and are dominated by communities generally comprised of salt-tolerant herbs, shrubs, and grasses that have adapted to such dynamic environmental conditions (Nybakken 2001). In South African estuaries, salt marshes represent the most extensive primary producer habitat (Adams et al. 2026a), highlighting their ecological importance and significant contribution to estuarine productivity and ecosystem functioning.

Salinity across salt marsh habitats varies greatly depending on factors such as tidal stage and the amount of rainfall, and ranges both spatially and temporally from mildly brackish (1–10 ppt) to sea water (35 ppt) (Ashley et al. 2021). However, growth rates tend to decrease as salinity increases and germination occurs only when the surrounding water salinity is sufficiently low (Smart & Barko 1980, Price et al. 2011). The primary abiotic factors influencing salt marsh distributions are salinity and water availability (Pan et al. 1998). Salt marshes growing in areas with high water availability (high rainfall and intertidal zones) are influenced by sediment salinity more than by water availability in terms of zonation patterns (Krüger & Peinemann 1996). Sediment moisture limits the growth of xerohalophytes (salt-tolerant plants that occur in dry or mostly dry soils), which in turn is dependent on the depth of the water table (Zedler et al. 1986, Bornman et al. 2008).

Within traditional salt marshes, plant communities occur along distinct zones following tidal inundation and salinity gradients, along which different plant species and vegetation colours are seen (Noe & Zedler 2001, Álvarez Rogel et al. 2001, Hughes & Paramor 2004, Perry & Atkinson 2009). For example, salt marsh habitats are typically categorised as either intertidal or supratidal, with intertidal salt marsh occurring below the mean high water spring tide mark and supratidal salt marsh above this. As few species are able to cope in highly saline marine environments with fluctuating water levels, species diversity within marsh communities is typically low. However, South African salt marshes are species rich with communities often comprising up to 95 different plant species, far greater than the number of plant species found in salt marsh communities in Britain (45) and the United States (43) (Veldkornet et al. 2015).

Langebaan Lagoon and Knysna both contain extensive intertidal salt marsh areas due to their large tidal prisms and gentle slopes, with Langebaan Lagoon incorporating the second largest salt marsh area in South Africa, accounting for approximately 30% of this habitat type in the country (Tabot & Adams 2019, Adams et al. 2026b). The spiky cord grass *Spartina maritima* and perennial glasswort *Sarcocornia perennis* are highly abundant in the Lagoon, especially along the coast of Schrywershoek and Geelbek (Figure 12.2), and the intrusion of fresh groundwater into the Lagoon maintains large rush (*Juncus* spp.) stands.

Approximately 43% of South Africa's national salt marsh area has been lost since the 1930's (Adams et al. 2026a). It is estimated that 10 000 ha of salt marsh has been lost

due to agricultural expansion and development within the floodplain, including the construction of causeways, bridges, housing and business developments, footpaths, embankments, and jetties (Adams 2020). The remaining extent (~13 000 ha) continues to face multiple pressures, including salinization associated with reduced freshwater inputs, flooding of marsh habitat due to estuary mouth closure, altered hydrological regimes caused by increased storm surges, and climate change-driven sea-level rise and habitat replacement (Adams 2020, Adams et al. 2026a). The greatest loss of macrophyte habitat has been that of supratidal salt marsh, likely because it often forms the ecotone between salt marsh and terrestrial vegetation and is therefore the most likely habitat to be developed (Adams et al. 2026b).

Figure 12.2. (Top) Beds of spiky cord grass *Spartina maritima* near Schrywershoek. (Photo credit: Zachary Christensen) (Bottom) Perennial glasswort *Sarcocornia perennis* at Geelbek.

Despite widespread declines of this habitat type across South Africa, the salt marshes of Langebaan Lagoon have not exhibited a similar trend in recent years. Between 1937 and 2021, the extent of supratidal salt marsh in the Lagoon increased from 1 075 ha to 1 132 ha, while the area of intertidal salt marsh remained largely unchanged As a result, Langebaan Lagoon represents the only documented historical increase in salt marsh extent in a South African estuary in recent decades (Adams et al. 2021). This increase has been attributed to the establishment of a protected area managed by SANParks, which reduced pressures associated with livestock grazing and agricultural activities (Adams et al. 2021).

12.1.3 COMMON REED

The southern-most section of Langebaan Lagoon (furthest from the Bay, near Geelbek) supports unique wetlands supported by the intrusion of fresh ground water, rather than by overland streams or rivers. The Geelbek area is a significant discharge zone for the Elandsfontein Aquifer system, and it has been suggested that the groundwater entering the Lagoon plays a significant role in sustaining the marsh ecosystems at the head of the lagoon (Valiela et al. 1990, Burnett & Stickle 2001).

This area contains reed beds dominated by common reeds Phragmites australis (Figure 12.3) and Cape bulrushes Typha capensis. These are freshwater-associated plants, meaning they can only survive in water or, at minimum, damp soil, and once established can tolerate salinity levels up to a maximum of 20–25 ppt at which point juvenile plants experience high mortality rates (Karberg 2015). The salinity of the water in the Lagoon is generally the same (or occasionally higher) as that of seawater (34–35 ppt). Consequently, these species can be used as "diagnostic plants" as their presence confirms that groundwater inputs are influencing the chemistry of the Lagoon (Christie 1981, Allanson & Baird 1999, Hockey & Turpie 1999, Whitfield 2019).

Figure 12.3. Common reed *Phragmites australis*.

Reeds, sedges and rushes (17 530 ha) are the dominant estuarine habitat type in South Africa (Adams et al. 2026b). This habitat is increasing in size across the country due to the nutrient enrichment of estuaries which encourages reed growth, although the cover of reeds is highly dynamic, with dieback occurring in winter and during flooding events (Adams et al. 2026b).

These areas form important habitats for birds, invertebrates and fish, and food for detritivores in the fresh and brackish water zones of estuaries. They perform important ecosystem services as they, like salt marshes, protect the banks of estuaries from erosion and are important for carbon storage (Adams et al. 2026b). Reeds can also help improve water quality, as their stalks trap sediment and absorb excess nutrients and phosphorous. This, paired with their high productivity, is why reeds like *Phragmites* are

widely used in the construction of artificial wetlands for the treatment of polluted water (Adams et al. 2026b).

The Kropz Elandsfontein Phosphate Mine, located approximately 11 km to the east of Langebaan Lagoon, extracts and uses groundwater for its mining operations, before reinjecting it back below the water table. Although this technique in theory should protect the local ground water resources, and it is unlikely to have any impact on the groundwater quality and flow, it is possible that this operation could lead to a reduction in freshwater flows. This would place significant pressure on common reed populations in the lagoon, making this threat a key one to monitor in the context of this long-term monitoring program of Saldanha Bay.

12.1.4 Historic monitoring efforts of macrophytes

Past macrophyte research and monitoring programmes in South Africa and within Langebaan Lagoon primarily relied on traditional in-situ monitoring techniques for measurements of macrophyte extent and condition. In most cases, a small project focal area was necessary as it reduced the cost of sampling efforts, while addressing a select topic of concern. These programmes were successful in identifying localised patterns and changes to the extent and condition of various macrophytes, primarily seagrasses. However, it remains important that long-term monitoring continues to build upon historic research while informing the stakeholders of the current condition and extent of these keystone species.

As part of the 2021 State of the Bay (SOB) monitoring programme, a geospatial workflow was developed to leverage advances in remote sensing for assessing changes in Langebaan Lagoon's coastal vegetation. The workflow has subsequently been refined during the 2023 and 2025 monitoring cycles using Google Earth Engine and ArcGIS Pro to map the variability in extent over time of macrophyte communities through supervised image classification. Supervised image classification schemes were developed to map the extent of multiple macrophyte classes including seagrass, salt marsh, and common reed. This approach groups pixels with similar characteristics to delineate habitat classes, including seagrass, salt marsh, and common reed. Prior to 2026, the seagrass and salt marsh classifications provided unvalidated (i.e. without ground-truthing) estimates of the extent and variability of non-species-specific habitats. However, the classifier developed for common reed is trained by GPS tracks drawn in 2020 around known populations of *P. australis* at Geelbek and is therefore partially (historically) validated.

In 2025, a new approach to supervised classification based on spectral signature analysis was implemented to improve the accuracy and consistency of remote sensing-based habitat mapping. Because surface materials such as vegetation, water, bare sediment, and built infrastructure absorb and reflect different wavelengths of light in characteristic ways, their unique spectral signatures can be used to quantitatively distinguish features and improve image classification. The incorporation of spectral signature analysis substantially improved the classifier's ability to discriminate between submerged vegetation types, which exhibited distinct reflectance characteristics across Sentinel-2 bands 4–9 (650–960 nm), encompassing the red, red-edge, and near-infrared regions of the electromagnetic spectrum. The distinct spectral responses observed across these bands provided the foundation for a more robust supervised classification

workflow. The methods used to develop the spectral signatures and implement the classification approach are described in the following section.

Although the classification workflow has been progressively refined since 2021, the principal methodological advancement in 2026 is the introduction of preliminary field validation for submerged vegetation. Rather than a comprehensive ground-truthing exercise, opportunistically collected photographs and vegetation samples were identified under laboratory microscopy to provide an initial verification of the seagrass classification. The following section describes the methods used to acquire these observations and incorporate them into the supervised classification workflow.

12.2 METHODS

The 2026 assessment employed a standardised and reproducible geospatial workflow to map the distribution and extent of macrophyte communities in Langebaan Lagoon. The methodology integrates remote sensing, spectral signature analysis, supervised image classification, and targeted field observations to support consistent comparisons between monitoring years.

12.2.1 SATELLITE IMAGERY AND PRE-PROCESSING

The Copernicus Sentinel-2 satellite constellation provides freely available multispectral imagery well suited to vegetation and habitat mapping applications. With a spatial resolution of up to 10 m and 13 spectral bands spanning the visible, red-edge, near-infrared, and shortwave infrared regions of the electromagnetic spectrum, Sentinel-2 imagery provides the spatial and spectral detail required to distinguish coastal vegetation communities and monitor changes in their extent (ESA 2015). The constellation's high revisit frequency (5–6 days) also enables the acquisition of cloud-free imagery throughout the year, supporting seasonal assessments as well as monitoring before and after major environmental events.

Sentinel 2 Level-2A imagery (COPERNICUS/S2_HARMONIZED) covering Saldanha Bay and Langebaan Lagoon was processed using Google Earth Engine (Gorelick et al. 2017). Images acquired during May to July of each year between 2015 and 2026 were selected because this period consistently provided the greatest availability of cloud-free imagery and the strongest spectral differences among coastal vegetation types, associated with the transition from the dry to the rainy season. Before classification, the imagery was clipped to the study area using two predefined extents: a coastal assessment extent and a water extent (Figure 12.4). A permanent water layer, derived from the Global Surface Water dataset (Pekel et al. 2016) was used to separate terrestrial and aquatic analyses. This layer excluded permanent water from the mapping of coastal vegetation (common reed and salt marsh) while defining the processing extent for submerged vegetation (seagrass).

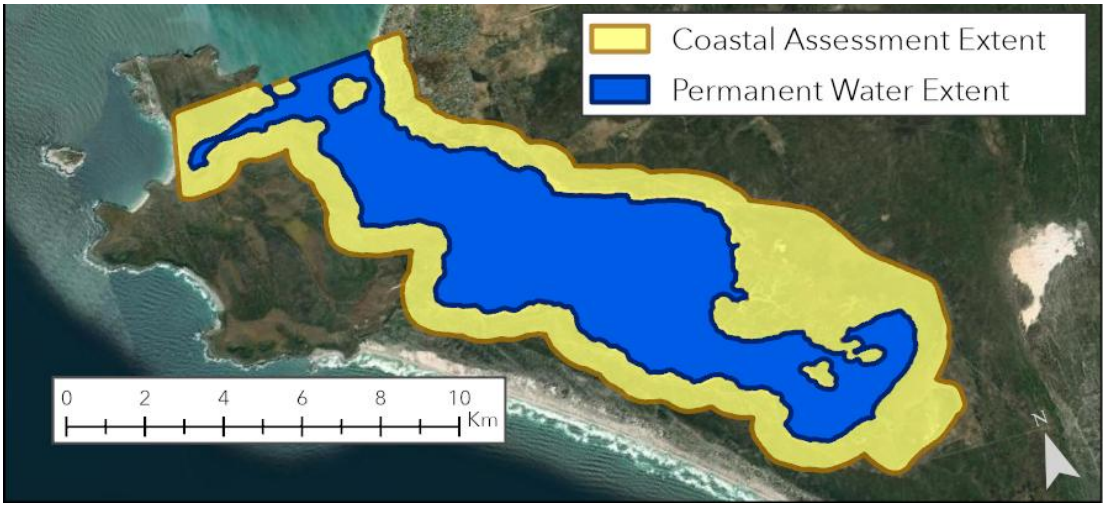


Figure 12.4. The analyses extent layers that were used to crop imagery prior to classification.

12.2.2 Supervised image classification

Supervised image classification using spectral profiling was applied to map the spatial extent of macrophyte communities within Langebaan Lagoon from processed Sentinel-2 imagery. This approach assigns individual pixels to predefined habitat classes by comparing their spectral characteristics with representative training samples developed for each vegetation type. Spectral signatures represent the unique reflectance patterns of surface materials across different wavelengths, which arise from differences in physical and biological properties such as pigment composition, water content, and structure.

Spectral profile charts visually display the reflectance values of selected areas of interest across all Sentinel-2 bands (Figure 12.5), enabling the development and refinement of representative spectral signatures for each macrophyte class. These signatures improved the separation of spectrally similar habitats, particularly submerged vegetation, where differences in reflectance across the red, red-edge, and near-infrared wavelengths supported the identification of seagrass habitats. Separate classification workflows were applied to aquatic (seagrass) and terrestrial (salt marsh and common reed) habitats using the corresponding processing extents described above.

12.2.3 Opportunistic validation (seagrasses)

Classified imagery from the 2025 programme was reviewed and used to guide opportunistic field verification during the 2026 monitoring campaign. Classification outputs were used to identify representative submerged vegetation areas for targeted site visits, where seagrass observations were collected through photographic records and vegetation samples. These samples were subsequently identified in the laboratory to provide preliminary validation of the remotely sensed seagrass classification and support further refinement of the classification workflow. This, alongside future in situ validation-based approaches will contribute to increasing the accuracy with which different macrophyte classes are mapped. Additionally, various epiphytes found on the macrophyte samples were observed under dissecting microscopes and identified to the lowest taxonomic level possible using field guides such as the Two Oceans: A Guide to the Marine Life of southern Africa.

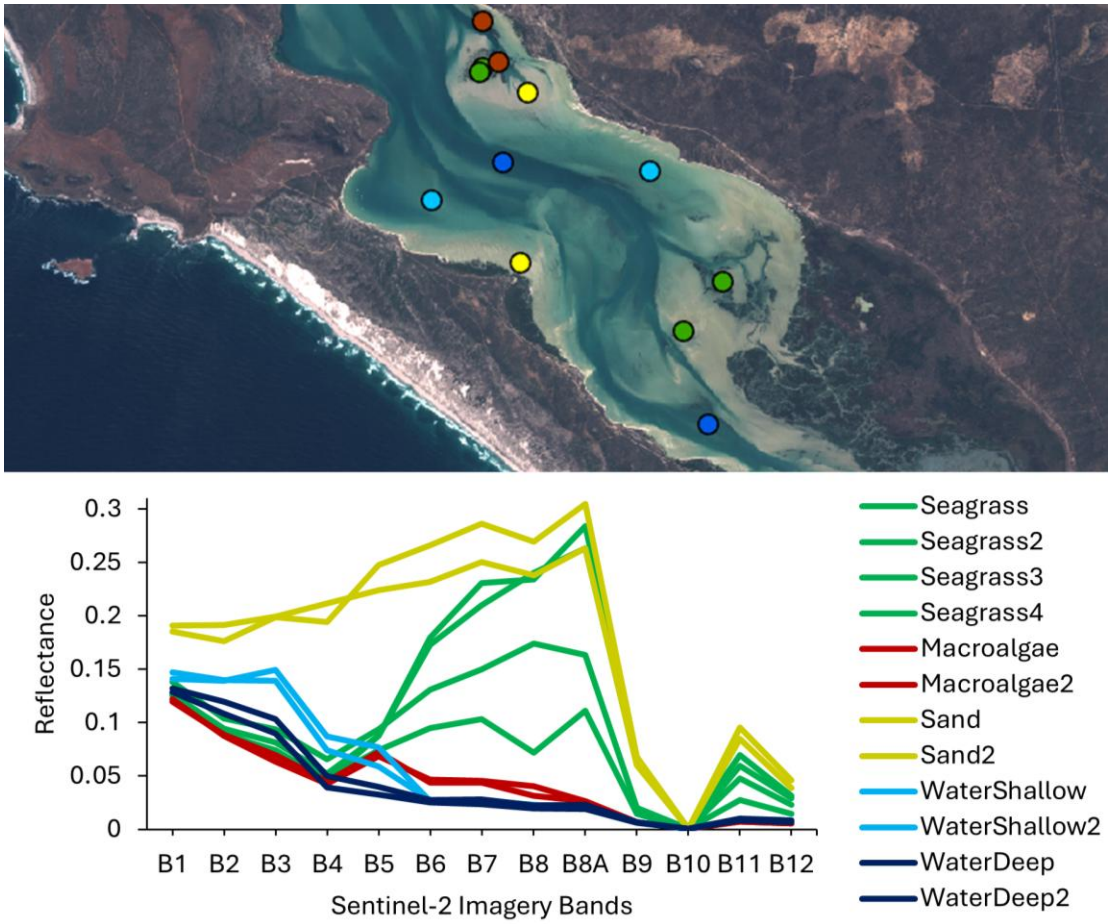


Figure 12.5. Spectral analyses of points placed within various aquatic classes (dark blue: water channels, light blue: shallow water, yellow: sand flats, green: seagrass, red: submerged macroalgae).

12.2.4 NET CHANGE ANALYSES

Net change analysis is a spatial assessment approach used to quantify changes in habitat extent over time by comparing the presence, loss, and persistence of a habitat class between multiple monitoring periods. In this application, the classified macrophyte maps were used to evaluate temporal changes in distribution by identifying areas of habitat persistence, expansion, and loss across the monitoring period (2015–2026). The classification process produced habitat maps where each pixel was assigned to a specific class based on its spectral characteristics. These outputs were refined by removing incorrectly classified features, such as open water and sand, to create simplified maps representing the presence or absence of each macrophyte type. Areas classified as present were converted into mapped features, allowing the total extent of each habitat type to be calculated for each year.

The annual presence-absence maps were then combined to identify spatial patterns of change over time (Figure 12.6). By comparing the overlap between yearly habitat maps, each location was categorised according to the frequency and persistence of macrophyte occurrence throughout the monitoring period. The resulting dataset was used to generate a summary of habitat stability and change, highlighting areas of long-term persistence, expansion, and decline in macrophyte extent.

For example, pixels classified as seagrass all five years between 2021 and 2025 (line b in Figure 12.6) are present in 2025 and were present in all images, indicating stable and unchanging seagrass extent. Pixels classified as seagrass for a single year in 2022 (line a in Figure 12.6) and are not present in the most recent image is indicative of transient or lost seagrass beds. On the contrary, pixels classified as seagrass for all years except the most recent is indicative of very recent loss of long-standing seagrass beds and may suggest a new source of disturbance was introduced in that area within the last 12 months. Once the reclassification is complete, the table is joined with the final net change raster and re-symbolised to reflect growth (green) and decay (red) colouration.

In the net change analysis outputs, red pixels represent areas where macrophyte habitat was present during previous monitoring years but was absent in the final year of analysis (2026) (Figure 12.7). Light red pixels indicate areas where macrophytes were detected for only one previous year, while progressively darker red pixels represent areas with a longer history of macrophyte presence. The darkest red pixels identify historically persistent macrophyte habitat that was lost between 2025 and 2026, representing priority areas for future monitoring due to recent changes that may have contributed to habitat decline.

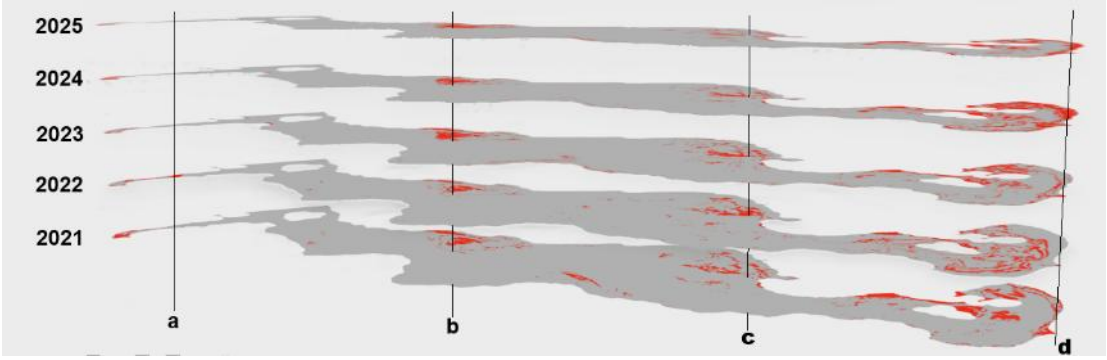


Figure 12.6. Visual representation of the combined raster process for seagrass in Langebaan Lagoon. These rasters are binary; grey is pixel value 0 representing no presence, and red is pixel value 1 indicating presence. The lines with the following letters are drawn as examples to depict pixel overlap and explain the resultant output pixel values assigned to each unique combination. (a) present for 1 out of 5 years, not present in 2025, (b) present for all 5 years, (c) present for 3 out of 5 years, not present in 2025, (d) present for 4 out of 5 years and present in 2025.

Light green pixels represent areas of recent macrophyte establishment or expansion relative to 2026, while dark green pixels indicate stable macrophyte habitat that was consistently classified as present throughout the entire analysis period. Together, these outputs highlight areas of habitat persistence, loss, and recent change, providing insight into long-term patterns of macrophyte stability and vulnerability.

Figure 12.7. Colour symbology of the net change rasters presented in the results.

12.3 Results and Discussion

The 2026 macrophyte monitoring results demonstrate considerable spatial and temporal variability in the distribution and extent of seagrass, salt marsh, and common reed habitats within Langebaan Lagoon. The results of the 2015–2026 supervised classification visualised by means of net change analyses are effective in distinguishing persistent habitat from areas of intermittent establishment and loss. The findings are considered in relation to observed patterns of habitat variability, known ecological drivers, and the limitations and uncertainties associated with satellite-based classification. It is recommended that particular attention be given to areas exhibiting persistent habitat loss or change, which may warrant further investigation and targeted ground-truthing.

12.3.1 Salt marsh distribution and net change 2015–2026

Reports that salt marsh extent in Langebaan Lagoon has remained fairly stable in recent decades (Adams 2020) are supported by the results of this year's classification effort (Figure 12.8). Areas of recent habitat loss (light red patches) are generally confined to these margins of existing marshes, suggesting the loss of habitat that was only intermittently present during the monitoring period. There are relatively few large areas of dark-red pixels, indicating that the loss of long-standing salt marsh habitat has been limited. In fact, there was a marked increase in classified extent between 2024 and 2026 (Figure 12.8), suggesting either improved environmental conditions favouring salt marsh expansion, interannual variability in vegetation detectability, or ongoing refinement of the classification model. Additionally, the supervised classifier detected persistent salt marsh habitat in Rietbaai, between Preekstoel and Churchhaven, in Boerplein, Schrywershoek, at the head of the Lagoon in Bokramsloot and Geelbek, and most extensively between Geelbek and Bottelary. The classifier detected stable, unchanging salt marsh extent covering 205.83 hectares for the years of 2015–2026 (dark green in Figure 12.10).

While the core persistent areas of the salt marsh habitat are classified consistently throughout the 12-year period, the edges of the habitat exhibit substantial spatial variability from year to year (Figure 12.10). Areas of habitat turnover are usually concentrated along marsh edges, where vegetation expands and contracts in response

to changing environmental conditions. Most of the core persistent (dark green) patches of salt marsh are fringed by areas of recent establishment on the landward side (light green) and areas of loss on the lagoon-facing side (light red). This broad pattern suggests a gradual shoreward migration of the salt marsh boundary, with habitat being lost along the seaward margin and simultaneously colonising suitable area further inland. This may be in response to factors such as sea-level rise, changes in sediment dynamics or changes in hydrology (Donnelly & Bertness 2001).

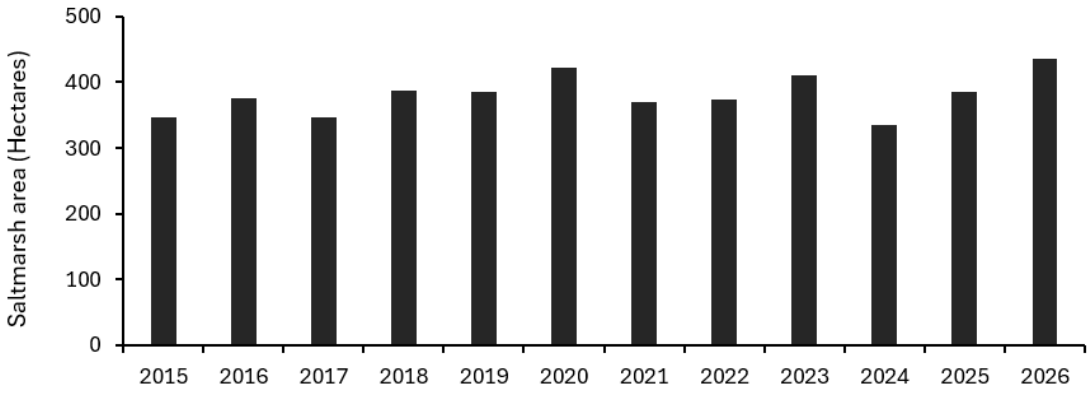


Figure 12.8. Change in the spatial extent of salt marsh in Langebaan Lagoon over time (2015–2026).

12.3.2 SEAGRASS DISTRIBUTION AND NET CHANGE 2015–2026

The 2026 net change analysis of submerged macrophytes in Langebaan Lagoon indicates that seagrass distribution remains highly dynamic (Figure 12.11). There are relatively small cores of long-term permanent meadows at Centre Banks and to the southwest of Bottelary, with the classifier detecting the same 5.44 ha of seagrass every year between 2015–2026 at Centre Banks meadow (towards the centre of the Lagoon between Oesterwal and Shark Bay). Centre Banks and the area west of Bottelary are shallow (0–2 m depth), sheltered intertidal areas characterised by fine sands and muddy sediment. The persistence of these meadows suggests that environmental conditions at these sites, including water depth, sediment stability and light availability, remain favourable for long-term seagrass survival.

The classified seagrass extent varied considerably between 2015 and 2026, with a general declining trend following a maximum extent of approximately 300 ha in 2019 (Figure 12.9). Between 2015 and 2018, seagrass extent fluctuated between 240 and 280 ha, before increasing to its highest recorded extent (~300 ha) in 2019. Extent remained relatively high in 2020 and 2022 but subsequently declined to 228 ha in 2023 and a minimum of approximately 200 ha in 2025. In 2026, the classified extent increased to approximately 227 ha, representing a recovery of approximately 28 ha from the previous year. Despite this increase, the 2026 extent remains approximately 75 ha below the 2019 maximum, indicating that the observed recovery has not yet returned seagrass extent to the levels recorded during the earlier part of the monitoring period.

Transient and patchy seagrass meadows were detected throughout the shallows of the majority of the Lagoon. Towards the edge of the permanent water feature, seagrasses form ecotones with coastal macrophytes, often growing in mosaics with salt marshes

and terrestrial grasses. This is detected especially along the coasts of Rietbaai, Klein Oesterwal to Oesterwal, Bottelary, and throughout the head of the Lagoon at Geelbek (Figure 12.11). Due to the patchy, clustering formation of seagrasses which often only form meadows between 1–5 m^2, and the spatial resolution of the Sentinel 2 imagery, the classifier cannot detect smaller meadows.

These results also show considerable temporal variability. Large areas of the lagoon are covered in light green and light red classes, indicating that seagrass presence and distribution frequently shift between years rather than remaining permanently established in a few concentrated locations. This pattern is consistent with the ecology of *Zostera capensis*. This species has been shown to exhibit pronounced seasonal diebacks in Langebaan Lagoon, particularly in late summer, followed by regrowth in autumn (Lawrence 2025). This seasonal dieback strategy relies on reserves in belowground tissues to regenerate aboveground biomass when conditions are favourable. Local-scale environmental conditions, such as tidal exposure, turbidity and water retention, also influence seagrass dynamics and may contribute to the high levels of natural dynamism in their distribution on the lagoon.

Red pixels (areas where seagrass was detected in previous years but was absent in 2026) are widespread throughout the lagoon, with most losses occurring as narrow bands around meadow edges and close to the shoreline. This likely reflects interannual shifts in meadow boundaries and biomass due to changes in local scale environmental variables or limitations associated with detecting small and fragmented patches of seagrass using coarse resolution satellite imagery. However, several dark patches indicate the loss of historically persistent seagrass areas, especially around Geelbek, Rietbaai and parts of Centre Banks. These locations should be considered priority monitoring areas as these losses may be related to localised environmental change or increasing anthropogenic pressure.

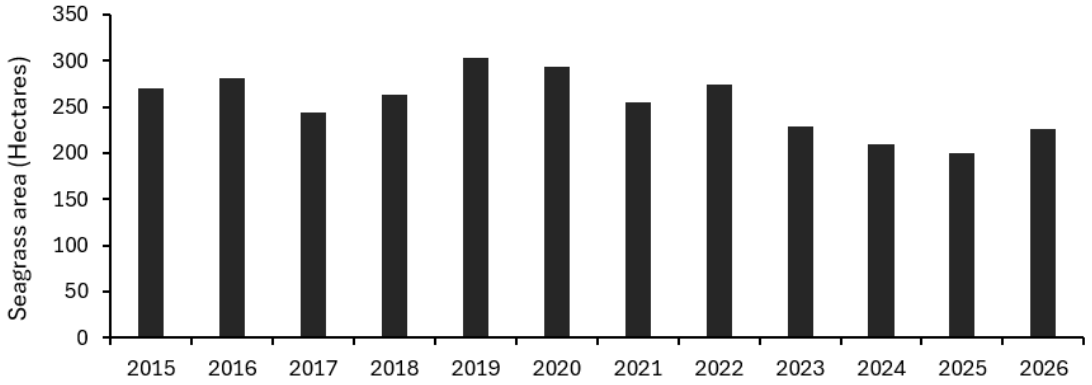


Figure 12.9. Change in the spatial extent of seagrasses in Langebaan Lagoon over time (2015–2026).

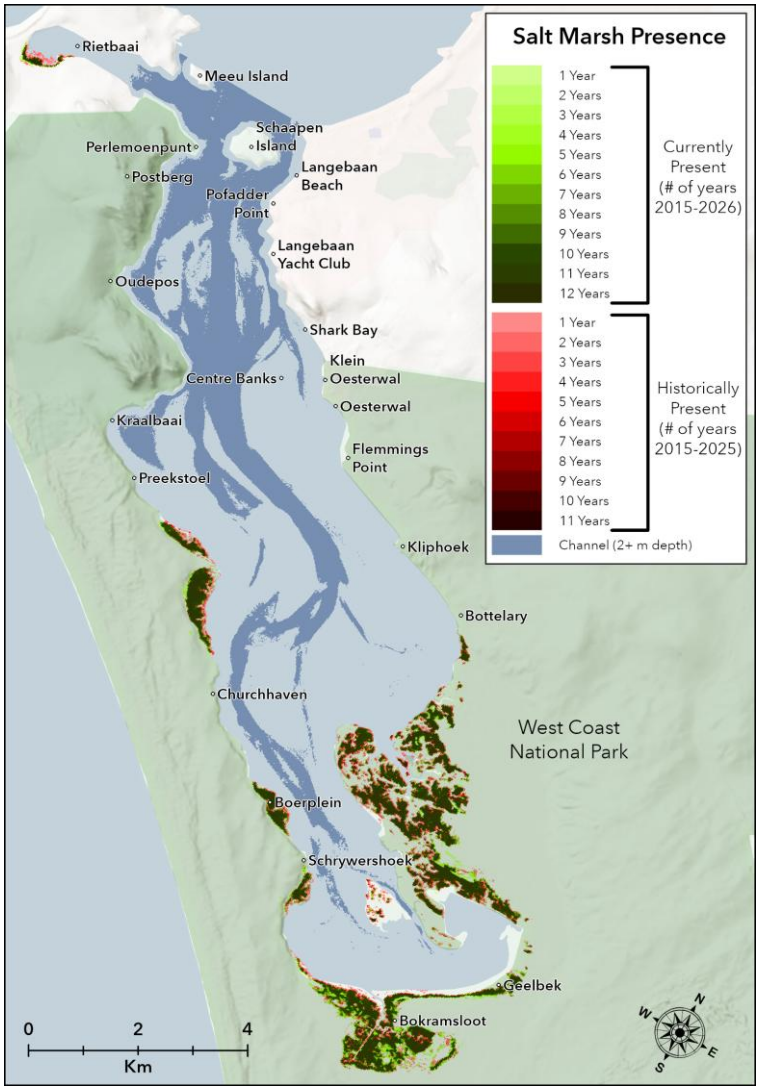


Figure 12.10. Spatio-temporal variability of salt marshes in Langebaan Lagoon (10 m spatial resolution, 2015–2026).

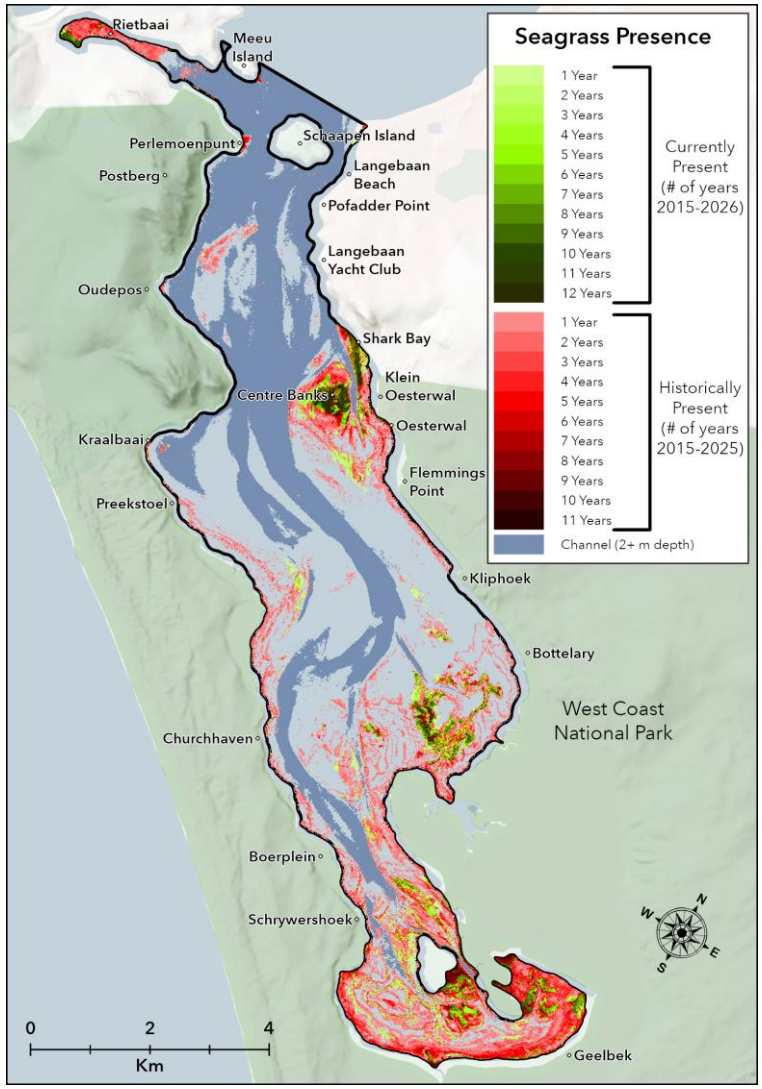


Figure 12.11. Spatio-temporal variability of seagrass beds in Langebaan Lagoon (10 m spatial resolution, 2015–2026). Dark blue represents a water channel that is deeper than the intertidal zone.

12.3.3 Common reed distribution and net change 2015–2026

In line with previous unsupervised (2020) and supervised (2021, 2023, 2025) classification efforts of common reed, the supervised classification effort in 2026 targeted the southern and south-eastern shores at the head of the Lagoon, near Geelbek, where common reed dominates the flora of the reedbeds where groundwater inflow occurs. These reeds are visible from high resolution satellite imagery (Figure 12.12).

Figure 12.12 A satellite image showing the stands of common reed (circled in red) at the head of Langebaan Lagoon, near Geelbek.

The classified common reed extent remained relatively stable over the 2015–2026 monitoring period, with annual estimates generally ranging between approximately 70 and 91 ha (Figure 12.13). Extent increased marginally from 81 ha in 2015 to 86 ha in 2017, before declining to approximately 71 ha in 2018. This was followed by a recovery to approximately 86 ha in 2019 and 83 ha in 2020, with the greatest extent recorded in 2021 at approximately 91 ha. A subsequent decline was observed in 2022 and 2023, reaching a minimum of approximately 74 ha in 2023. Since then, classified extent has increased consistently, reaching 87 ha in 2026, the second-highest extent recorded during the monitoring period.

The net change visualisation depicts a large contiguous area of dark green pixels concentrated along the eastern shoreline and extending inland from the lagoon margin. These pixels represent reed beds that were present throughout most or all of the monitoring period (10–12 years), indicating the presence of a stable and persistent reed population. The stable, unchanging common reed extent (darkest green, Figure 12.14) covered 33.52 ha for the years of 2015–2026.

The pattern of light red and light green cells surrounding the large contiguous areas of dark green cells indicate natural fluctuations at the edges of the distribution. These changes may be attributed to factors such as local shifts in inundation patterns, sediment redistribution, salinity fluctuations (likely linked to groundwater or rainfall

changes), seasonal variability, or the inherent uncertainty associated with mapping narrow habitat boundaries at a 10 m spatial resolution. There are several darker red patches, particularly in the northern section of the common reed stand. These areas (locations that supported reeds for many consecutive years prior to disappearing in 2026) are potentially more significant than the losses observed at the dynamic margins, as they may indicate recent environmental change affecting established stands. These locations should be considered priority areas for future ground-truthing and investigation to determine whether changes in freshwater inflow, groundwater availability or local disturbance have contributed to the observed decline.

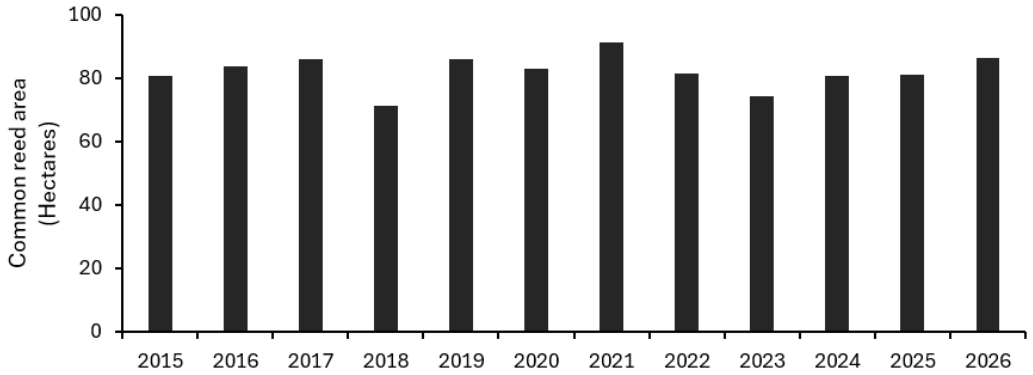


Figure 12.13. Change in the spatial extent of common reed at Geelbek over time (2015–2026).

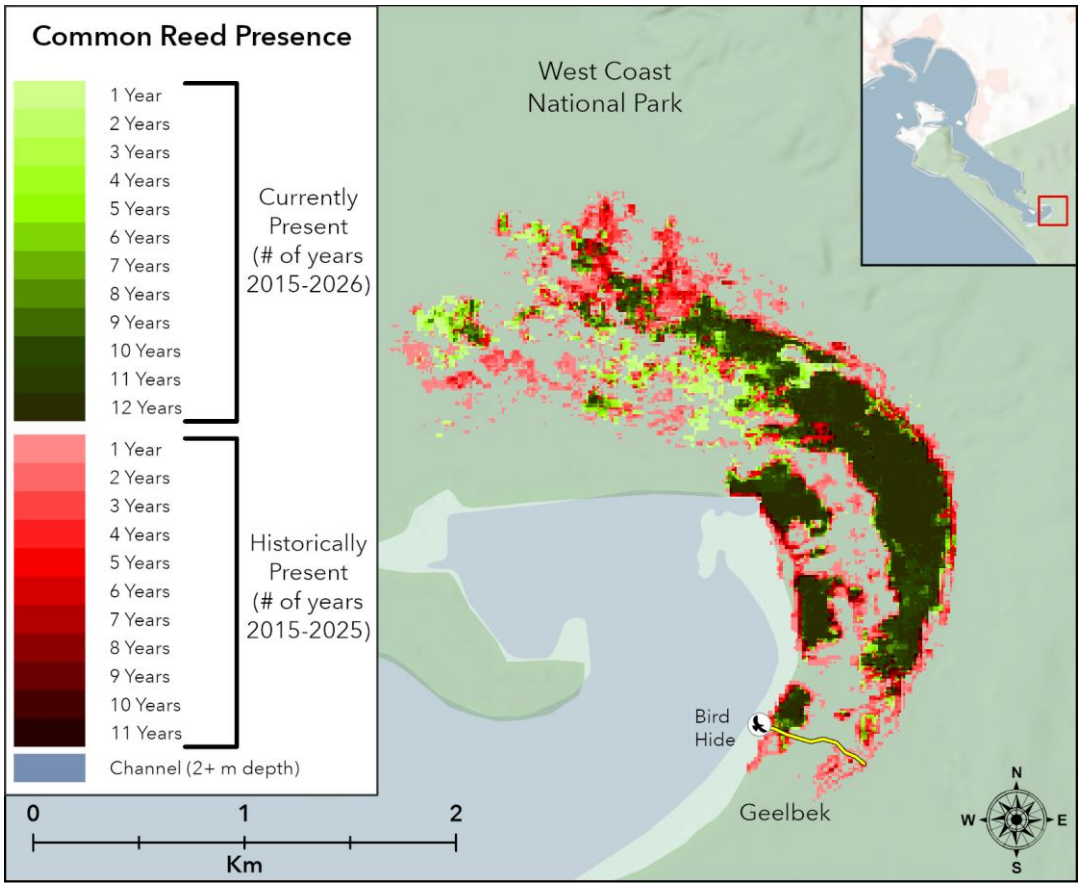


Figure 12.14. Spatio-temporal variability of common reed at Geelbek, head of Langebaan Lagoon (10 m spatial resolution, 2015–2026).

12.3.4 OPPORTUNISTIC VALIDATION (SEAGRASS)

Opportunistically collected vegetation samples and georeferenced photographs were analysed to support the supervised classification analysis. In the process of distinguishing between macroalgae samples and seagrass samples, it was discovered that what at first appeared to be macroalgae was in fact seagrass vegetation covered in macroalgal epiphytes (Figure 12.16). Epiphytes–organisms that grow on top of plants–were found on most of the seagrass samples collected from Langebaan Lagoon in 2026 (Figure 12.15), with one of the epiphytic species identified as fine red algae from the genus *Ceramium*. The photographs taken (Figure 12.16) suggest that epiphyte growth on seagrass plants in the lagoon is high.

This observation has important implications for the interpretation and accuracy of the supervised classification results. The presence of epiphytes can substantially alter the spectral reflectance of seagrass by introducing the spectral characteristics of the epiphytic organisms to those of the underlying seagrass leaves. Where epiphyte cover is extensive, the resulting spectral signature may therefore resemble that of macroalgae more closely than that of relatively clean seagrass. This creates a potential source of confusion between the two classes, particularly where macroalgae and epiphyte-covered seagrass occur in similar shallow-water environments. Consequently, areas of seagrass with high epiphyte loads may be incorrectly classified as macroalgae or excluded from the seagrass class, resulting in an underestimation of seagrass extent. Conversely, if epiphyte-covered seagrass is incorporated into the seagrass training samples without being recognised as such, the classifier may develop a broader seagrass spectral signature and potentially increase the likelihood of classifying spectrally similar macroalgae as seagrass.

Figure 12.15 A *Zostera capensis* sample collected in Langebaan Lagoon covered in epiphytes, with photographs of the epiphytic fine red algae (*Ceramium* spp.) under the microscope.

The degree of this effect is likely to vary spatially and temporally depending on the abundance, composition and condition of epiphytes, as well as water depth and other physical factors influencing the spectral signal interpreted by the classifier. This introduces an additional source of uncertainty when comparing classified seagrass extent between years, particularly where epiphyte abundance varies between image acquisition dates. The 2026 observations therefore highlight the importance of distinguishing between seagrass presence and the spectral condition of the seagrass canopy when developing training data. Future monitoring should incorporate targeted field validation of both relatively clean and epiphyte-covered seagrass, together with representative macroalgae samples, to better characterise the degree of spectral overlap between these classes. This would improve the ability of the classifier to distinguish seagrass from macroalgae and provide greater confidence in estimates of interannual changes in seagrass extent.

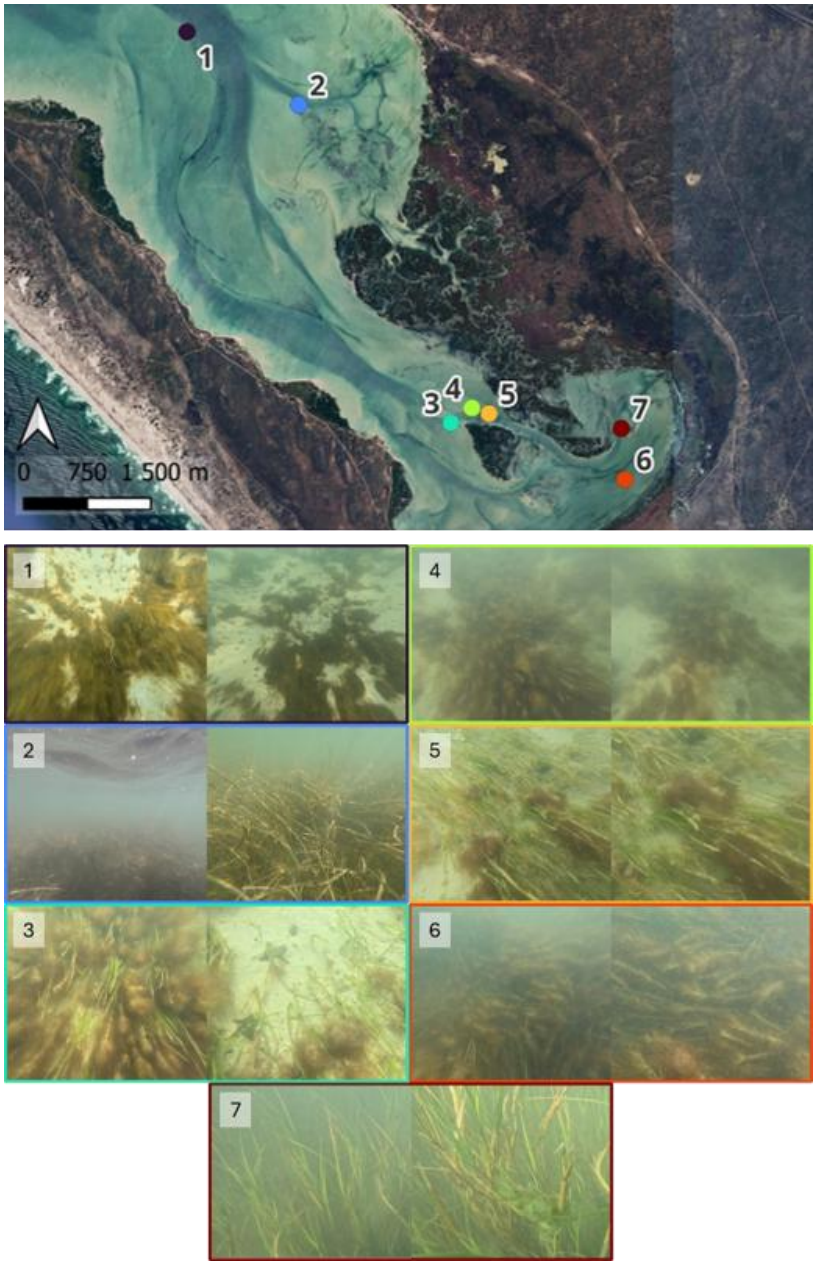


Figure 12.16 Photographs of seagrass (2-6), filamentous brown macroalgae (1) and grasses (7) at various locations within Langebaan Lagoon.

12.4 Discussion

The interannual variability observed in classified macrophyte extent reflects the dynamic nature of these habitats within Langebaan Lagoon but should also be considered in the context of limitations associated with satellite-based habitat classification. For seagrass in particular, differences in detectability and classification performance between years may contribute to some of the observed variability. The refined classification approach used in 2026, together with the incorporation of opportunistically collected ground-truthing data, may have improved the detection of smaller or previously unresolved seagrass patches. Consequently, changes in classified habitat extent should not necessarily be interpreted as direct evidence of equivalent changes in ecological condition or biomass.

The observed variability in seagrass extent is consistent with the naturally dynamic and patchy distribution of *Zostera capensis* within the Lagoon. Recent studies have focused a great deal on the structural and environmental drivers influencing seagrasses and their associated macrofaunal communities in Langebaan Lagoon (Lawrence 2024, 2025). For instance, the position of a seagrass population relative to the intertidal zone has a noticeable impact on seagrass, specifically the size of the leaves. Large-leaved seagrasses are more commonly found at the lower edge of the intertidal zone while small-leaved populations are found at higher shore positions (Lawrence 2024). This has ecological implications as different morphotypes support different macrofaunal communities, with large-leaved stands offering greater habitat complexity and relief from predation and therefore support a higher species diversity and richness than small-leaved stands (Lawrence 2024). Importantly, shallow intertidal sites like Geelbek and Bottelary are dominated by South Africa's most threatened marine invertebrate, the limpet *Siphonaria compressa* that primarily feeds on bacteria, diatoms and filamentous algae found on seagrass leaves (Lawrence 2024). As many as 100 individuals/ m^2 have been recorded at these sites relative to <5 individuals/m^2 at Oesterwal, and none at Klein Oesterwal and Centre Banks (Lawrence 2024).

The temporal variability observed in the classification is also likely influenced by the strong seasonal and localised environmental controls on macrophyte population growth and decay. Seagrass abundance is generally higher during spring and summer, while epiphytic and macrofaunal diversity and richness increase during autumn and winter (Lawrence 2024). Salt marshes undergo cycles of lateral expansion and erosion, resulting in changes in the position of the vegetation boundary over time. These dynamics are influenced by factors including tidal inundation, sediment dynamics and hydrology, and are likely to contribute to the greater variability observed along habitat margins compared with relatively stable core areas. While selecting satellite imagery for the period of May–July (autumn) each year allows for a clearer distinction between stands of salt marsh and common reed, and the surrounding vegetation, this likely causes a slight misrepresentation in the extent of stable seagrass stands given its significant seasonal variability.

These uncertainties are particularly important when considering the longer-term status of seagrass within Langebaan Lagoon. Previous studies have documented substantial historical declines, including multiple major reductions in seagrass beds at Klein Oesterwal and Bottelary between 1972 and 2004, and a 38% loss of *Z. capensis* at Klein Oesterwal, Oesterwal, Bottelary and Centre Banks between 1960 and 2007 (Angel et al. 2006, Pillay et al. 2010). Corresponding changes were also observed in densities of

benthic macrofauna at these sites, with species that were commonly associated with Zostera beds such as the starfish *Parvulastra exigua*, the limpets *S. compressa* and *Fisurella mutabilis* and general surface dwellers such as the gastropods *Assiminea globules*, *Littorina saxatilis*, and *Hydrobia* sp. declining in abundance. Species that burrowed predominantly in unvegetated sand, such as amphipods *Urothoe grimaldi* and the polychaetes *Scoloplos johnstonei* and *Orbinia angrapequensis* increased in density over that same period. Pillay et al. (2010) were also able to show that the abundance of at least one species of wading bird, the Terek Sandpiper *Xenus cinereus* (which feeds exclusively in *Z. capensis* beds) was linked to changes in the size of these beds, with population crashes in this species coinciding with periods of lowest seagrass abundance at Klein Oesterwal. By contrast, populations of wader species that do not feed in seagrass beds were more stable over time.

The estimated seagrass extent reported in this study (~200–300 ha between 2015 and 2026) is substantially greater than the 85.8 ha of submerged macrophytes mapped by Van der Linden (2014). Direct comparison between these estimates should be undertaken with caution because of major methodological differences. Van der Linden (2014) analysed aerial photographs from the National Geospatial Information of South Africa using ArcGIS 10.2 to delineate submerged macrophyte habitat (*Z. capensis*) from a single period (2013), whereas the present study employed Sentinel-2 multispectral imagery, supervised classification techniques, spectral signature analysis and annual mapping from 2015–2026. Differences in image source, classification approach, habitat definitions, mapping extent and the treatment of patchy or transient seagrass beds are all likely to contribute to this discrepancy. While some real ecological recovery or expansion of seagrass habitat may have occurred since 2013, it is more likely that a substantial proportion of the observed difference reflects methodological variation rather than a direct increase in seagrass extent.

Biological interactions may also influence the persistence and condition of macrophytes, particularly seagrass meadows. Epiphytes are an integral component of seagrass ecosystems and contribute significantly to primary production (Berlinghof et al. 2022). However, certain conditions, such as warming and nutrient enhancement, may promote excessive macroalgae and epiphyte growth on seagrass leaves (Burkholder et al. 2007) which compete with seagrass for sunlight (Borowitzka et al. 2006) and nutrients (Williams & Ruckelshaus 1993), and hinders oxygen and heat exchange between seagrass leaves and the surrounding water column (Noisette et al. 2020). Together, this can result in reduced rates of photosynthesis and a reduction in the overall fitness of seagrass plants (Michael et al. 2008, Noisette et al. 2020). While it is still unclear whether conditions in Langebaan Lagoon are causing epiphytes to have a significant negative effect on seagrass meadows, photographs taken in the Lagoon in 2026 suggest that epiphyte growth is high (Figure 12.16).

The invasive bryozoan *Amathia verticillata* represents a further potential pressure on *Z. capensis*. Recently introduced to Saldanha Bay, likely through vessel fouling, the species has subsequently spread into Langebaan Lagoon through tidal exchange and has been recorded within seagrass meadows (Figure 12.17). Surveys conducted in 2024 detected *A. verticillata* in 60% of sampled areas, including seagrass habitats, with 2 333 colonies recorded (Ackland et al. 2025a). The species has been documented in seagrass beds elsewhere and can cause meadow mortality through shading and fouling (Williams 2007). Its presence therefore represents an emerging pressure that should be considered

alongside the physical and environmental drivers of seagrass variability. The recent discovery of *A. verticillata* brings the number of known marine alien species in Saldanha Bay to 47, highlighting the pressure that alien invasive species are having on this ecosystem type.

Figure 12.17. Left: A large colony of *A. verticillata* smothering *Z. capensis* seagrass at low tide. Right: *A. verticillata* colonies anchored to sand prawn *Kraussillichirus kraussi* holes (Ackland et al. 2025a).

Anthropogenic disturbances, such as increased siltation from the bay due to dredging, boating activity, physical trampling by people engaged in water sports and climate change-related increases in water temperatures continue to place increased pressure on macrophyte ecosystems. While the precise reasons for the loss of *Z. capensis* beds remain speculative, the impact of human disturbance cannot be discounted, particularly at Klein Oesterwal where bait collection and in the last decade, kite surfing, has become very popular (Pillay et al. 2010). Most recent research in the Langebaan Lagoon shows that seagrass morphometric growth patterns are mainly controlled by temperature, followed closely by turbidity as a proxy for light levels (Lawrence 2025). It was found that cooler temperatures and less tidal exposure time favour higher seagrass biomass than warmer more exposed areas. This finding could partly explain the distribution patterns in the lagoon as determined from aerial photography (University of Cape Town, Cloverly Lawrence, *pers. comm*. 2014).

The ecological significance of maintaining seagrass habitat extends beyond the vegetation itself. Fluctuating meadow cover and localised extinctions can reduce the resilience of *Z. capensis* populations in parts of South Africa, which already exhibit low genomic diversity in parts of South Africa. The seagrass populations in Langebaan Lagoon may be of particular conservation significance, as this is the only known location where two distinct genomic clusters of *Z. capensis* co-occur, warranting increased research attention, management, and protection (Phair et al. 2019). These distinct genomic clusters relate to two observed morphotypes: short, stunted seagrass on the muddy tidal flats at Geelbek which experience prolonged periods of aerial exposure and longer seagrass with higher biomass on the permanently submerged sandy banks at Oostewaal (Phair et al. 2019, Watson et al. 2024). It is vital we protect this intrapopulation genomic diversity of this keystone species, especially in the context of safeguarding this keystone species and its resilience to future global change (Phair et al. 2019, Watson et al. 2024).

The loss of seagrass meadows has been shown to have profound implications for the biodiversity associated with them, including the loss of ecosystem services, species richness and species diversity. There is evidence to show that areas where seagrass is

lost can undergo a state change from a seagrass bed community to an unvegetated sand flat, as was the case at Klein Oesterwal in 2007 (Pillay et al. 2010, Clark et al. 2023). Sandflat species such as the burrowing sandprawn *K. kraussi* turn over massive quantities of sediment, and once established, effectively prevents re-colonization by seagrass and the species associated with it (Siebert & Branch 2005, Angel et al. 2006). Although the long-term effects of the loss of seagrass are not yet fully understood, studies suggest that reduced seagrass bed coverage and the associated changes to macro-invertebrates may have cascading effects on higher trophic levels (Whitfield et al. 1989, Orth et al. 2006).

Similar methodological considerations apply to the classification of salt marsh habitat. The structural complexity and species composition of salt marshes, together with their occurrence across the intertidal zone, introduce uncertainty into satellite-derived estimates in the absence of comprehensive in-situ validation. This may partly explain discrepancies between the classified extent and published estimates of salt marsh area within Langebaan Lagoon. The current approach also relies on a 30 m permanent-water delineation to segment the imagery prior to classification, which may be too coarse to accurately represent the complex shoreline around areas such as Geelbek. This could result in an overestimation of seagrass and an underestimation of salt marsh extent in areas where the two habitats occur in close proximity. Refining the training data through targeted ground-truthing, together with a higher-resolution (<10 m) delineation of the high-water mark and spatial representation of tidal influence, would improve the separation and classification of these intertidal habitats.

In contrast to the more pronounced variability observed in seagrass extent, common reed cover remained relatively stable over the 2015–2026 monitoring period. The largest declines corresponded with the severe drought during 2015–2018, while a smaller decline was also observed in 2023. Overall, the limited interannual variation suggests that the core common reed population has remained relatively persistent, with shorter-term fluctuations potentially reflecting changes in freshwater availability, rainfall and other environmental conditions, as well as variability in classification performance. The concentration of long-term persistent reed habitat suggests that hydrological connectivity, freshwater availability and sediment characteristics in this part of the Lagoon have remained broadly unaffected by anthropogenic activity and promote reed establishment and persistence. From a management perspective, these persistent reed stands represent an important ecological asset, while areas of recent or sustained habitat loss may provide an early indication of environmental change and should therefore be prioritised for field-based assessment and continued monitoring.

Overall, the 2015–2026 classification results demonstrate that the three macrophyte habitats exhibit different patterns of spatial persistence and temporal variability. Seagrass is the most dynamic of the habitats assessed and is subject to a combination of strong environmental controls, biological interactions and anthropogenic pressures, while common reed appears comparatively stable and closely associated with areas of persistent freshwater influence. Salt marsh occupies a particularly complex intertidal environment in which classification uncertainty remains a significant limitation. Continued monitoring, supported by targeted ground-truthing and refinement of the classification methodology, will be important for distinguishing genuine ecological change from variation in habitat detectability and for identifying areas requiring more focused investigation.

13 MANAGEMENT AND MONITORING RECOMMENDATIONS

13.1 OVERVIEW

Continuously accelerating urban and industrial development is a major cause of fragmentation and loss of ecological integrity of remaining marine and coastal habitats in Saldanha Bay and Langebaan Lagoon. The challenge of addressing cumulative impacts in an area such as Saldanha is immense. The current and future desired state of the greater Saldanha Bay area is polarised, where industrial development and conservation areas 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.

This chapter provides an overview of the environmental condition of Saldanha Bay and Langebaan Lagoon based on the results of the 2026 monitoring programme, considered from a management perspective. It brings together the evidence presented across the environmental indicators assessed and consolidates the key monitoring and management findings and recommendations into a concise and accessible format for managers and other interested parties.

The chapter highlights notable changes, emerging trends and areas requiring continued attention, and identifies where ongoing monitoring and management interventions may be required to address environmental pressures and support the protection of the ecological and socioeconomic values of the Saldanha Bay and Langebaan Lagoon system.

13.2 ACTIVITIES AND DISCHARGES

Major activities acting on the Bay that are considered in the chapter include human settlements, developments, water and wastewater; dredging; fish factories; marine aquaculture; and shipping, ballast water discharges and oil spills.

Significant monitoring activities are required as part of the Saldanha Bay ADZ Environmental Authorisation. This includes annual chemical survey monitoring, ongoing maintenance of continuous monitoring sensors within the Bay, hard substrate/reef structure surveys in Big Bay, data collection for dispersion model validation, qualitative sampling of fouling organisms, monitoring of benthic macrofaunal communities, and ongoing phytoplankton community monitoring.

Management interventions are required to limit further degradation of the environment from the growing pressures and should focus on the following issues:

- 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 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 LPG, 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 EIAs. Various environmental management instruments have been proposed for the Greater Saldanha Bay Area (refer to Chapter 3: Management and Policy Development). If these management instruments are indeed implemented, measures for the conservation alongside rapid development of the Saldanha Bay area will be addressed more effectively.

- Wastewater recycling should continue as wastewater production increases in the area.
- 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 WQG for the Coastal Zone and any other legislative requirements. There is evidence of a deterioration in some water quality parameters over time (such as pH for the Langebaan WWTW), and investigations should be made to ascertain and rectify the cause.
- A flow gauge monitoring volume of wastewater entering the Bay via the Bok river should be installed and/or monitored.
- Consideration should be given to the effects of anthropogenic ocean noise when planning or proposing projects which would increase shipping activity or noise.
- Existing and any future increases in use of groundwater from the Langebaan Road and Elandsfontein Aquifers need to be considered very carefully.
- Declaration of Saldanha Bay and Langebaan Lagoon as a Special Management Area in terms of ICMA should continue to be pursued.

13.3 GROUNDWATER

To support effective groundwater management in the Saldanha Bay area, the Saldanha Bay Municipality, in collaboration with GEOSS, maintains a dedicated monitoring network at the municipal wellfields, complemented by an extended network of DWS monitoring boreholes in the surrounding area. The network has remained unchanged over the past year at 24 boreholes, providing comprehensive spatial coverage of both the Langebaan Road and Hopefield wellfields, as well as key sites such as the Geelbek boreholes near the southern tip of Langebaan Lagoon. Groundwater levels and basic water chemistry (electrical conductivity) continue to be monitored monthly, with data uploaded to the regional database on a quarterly basis. Long-term observations indicate that groundwater levels and water quality remain stable, demonstrating that current abstraction and other uses are not imposing significant stress on the aquifer systems; this is reinforced by a dedicated cross-section assessment of the Elandsfontein mine's dewatering-and-recharge system, which confirms that its influence remains localised around the mine, with no evidence of it reaching the paleochannel pathway to Langebaan Lagoon. Abstraction at both wellfields remained well within licensed volumes this year, though volumes were notably lower than in previous years following vandalism at both

wellfields in September 2025; timely repair of the affected infrastructure is recommended to restore normal operations. It is strongly recommended that this monitoring continue to ensure ongoing protection of both the groundwater resources and sensitive ecological areas, and to support the implementation of planned numerical flow models for improved groundwater management and scenario testing.

13.4 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 WQT has, over the last seven to eight 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 need to be expanded to other areas and need to be extended to include a range of other water quality parameters such as dissolved oxygen, turbidity, nutrients, chlorophyll a (as measure of phytoplankton production).

The DFFE have also been 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[7]) for species identification and enumeration of phytoplankton. Water samples are also used for size-fractionated chlorophyll analysis. These data have already provided some insights into the effects of bivalve mariculture on water quality but logistical limitations on the sampling frequency have precluded further data analysis (Dawson et al. 2024). It has been shown in similar studies that the low oxygen conditions associated with shellfish farming in Small Bay also affects nitrogen cycling, as seen by low nitrate (NO_3^-) levels in the cold bottom waters during autumn. Coupled with the higher incidence of severely hypoxic and in some cases anoxic bottom waters within the farmed areas of Small Bay demonstrates the need to exercise caution in increasing shellfish production in Small Bay (Pitcher et al.). Inclusion of the nitrate and other data to the State of the Bay (SOB) Monitoring Programme in the future would be welcomed.

The Mussel Watch Programme samples collected from the shore and port infrastructure has been routinely analysed for the several trace heavy metals. Exceedance of food safety limits for lead, cadmium and arsenic in bivalves collected for research and farmed bivalves throughout the Bay, points to the need for management interventions, as metal contamination poses a serious risk to the health of people harvesting mussels.

Metal contamination poses a very serious risk to the health of people harvesting mussels from the shore (shellfish are harvested and consumed by recreational and subsistence fishers from the shore of the Bay). It is vitally important that this monitoring continues in the future and that data are made available to the public. 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). It is imperative that this Mussel Watch Programme be continued but revamped and possibly

[7] This South African Shellfish Monitoring and Control Programme is published by the Deputy Director General: Fisheries Management as a measure to facilitate compliance with Regulation 73 of the Regulations in terms of the MLRA (Government Notice R1111 in Government Gazette 19205 dated 2 September 1998).

extended to cover other species as well (e.g., fish) as elevated trace metal concentrations within seafood is a human health concern.

Mussel Watch data complements trace metals concentration reporting in mussels and oysters from the mariculture farms in the Bay (DFFE). This monitoring should continue but the (temporal) sampling frequency should be increased.

Microbial indicators (*E.coli*) revealed encouraging results at most sites in 2026 for recreational use limits. With respect to mariculture, the relatively new Water Quality Guidelines for mariculture, that are more stringent than the previous guidelines for faecal coliforms (DFFE 2022), meant most sites were non-compliant, from both Small Bay (as expected) and even Big Bay. Historically, Big Bay has generally complied with mariculture water quality guidelines due to effective tidal flushing. In 2026, however, several sites showed a broader decline in microbiological water quality, characterised by persistently, but only slightly, elevated faecal coliform concentrations rather than isolated pollution spikes. This suggests a temporary increase in background faecal contamination, potentially associated with diffuse land-based runoff, stormwater inputs, sewer infrastructure issues, or other catchment-scale influences.

Given the current importance of mariculture operation to Saldanha Bay, it is imperative that whatever efforts have been taken in recent years to combat pollution by harmful microbes in Small Bay should continue to be implemented.

The revised guidelines for water quality in South Africa (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. Incorporating this into the SOB monitoring programme would be welcomed.

13.5 COASTAL EROSION

Over the last few decades, the monitoring of coastal erosion and nearshore hydrodynamics in Saldanha Bay and Langebaan Lagoon has received more attention, inspired by observations of rapid coastal development and mounting concerns over the ecological health of the Bay. The monitoring effort for 2026 builds upon the long-term monitoring programme implemented in 2024 and is designed to show a more detailed analysis of coastal movement. The insights of this programme support management decision-making and help guide future mitigation efforts aimed at improving coastal stability.

Findings from 2025 and 2026 indicate sediment accretion has occurred primarily along southern, lagoon-facing sides of the groyne structures, while erosion has been observed along their northern sides and within the coastal compartment between the groynes. The analysis further indicates that the section between the groynes has undergone substantial shoreline retreat during the most recent five-year monitoring period. These results suggest that, although the groynes have slowed down the erosion rates and

promoted local sediment accumulation, they have not stabilised the entire Langebaan coastline or eliminated erosion within the wider coastal region.

Moderate correlations between shoreline change and wave dynamics further indicate that variations in wave climate contribute to shoreline variability. Consequently, these areas may be particularly vulnerable to future changes in coastal forcing, including more frequent or intense storm events and sea-level rise, which could exacerbate erosion and increase risks to nearby infrastructure.

The strong influence of wave dynamics suggests that future storms and sea-level rise may intensify erosion by increasing the amount of wave energy reaching the shoreline and allowing waves to extend farther landward. This is particularly concerning where the naturally protective calcrete outcrop continues to deteriorate following increased exposure to wave action and weathering. If appropriate measures are not implemented to preserve and stabilise the remaining beach areas and coastal stabilisation features at both Langebaan and Paradise beaches, continued coastal degradation is likely. This may further result in increasing risks to nearby infrastructure, recreational areas and public accessibility.

Although the present assessment has improved understanding of coastal vulnerability by examining the relationship between shoreline change and wave dynamics, further information is required to identify the full range of factors influencing coastal change. Therefore, future monitoring should incorporate more modern techniques such as unmanned aerial vehicle surveys along both beaches to provide higher-resolution measurements of shoreline and beach changes. Assessments should also include beach profiling, dune and vegetation-boundary mapping, beach geomorphology, sediment movement, sea-level rise, wind dynamics and individual storm events. These data would support the continued development of a comprehensive CVI, provide a finer-scale understanding of beach dynamics, improve assessments of current and future risks, and enable proactive management before more severe coastal damage occurs.

Furthermore, satellite data is limited by gaps in the revisit cycle and observations obscured by cloud cover. This makes it challenging to capture rapid (24–48 hour) changes to the coast, such as daily variability or those occurring immediately before and after storm events (seen in Section 7.4.2). To better understand these high-resolution coastal dynamics, more frequent and consistent monitoring of coastal movement is required. In recent years, community involvement in coastal research and management has evolved towards more active engagement in knowledge creation and decision-making processes. CoastSnap Community Beach Monitoring (CoastSnap) is a coastal citizen science program that leverages widespread smartphone technology alongside image processing techniques developed from decades of advances in optical remote sensing (Holman & Stanley 2007). CoastSnap employs the photo-point data collection method commonly used in other environmental monitoring efforts (e.g., Augar & Fluker (2014), NACC (2021)), which effectively engages a large number of opportunistic citizen scientists without requiring extensive volunteer commitment or training. Participants use a network of CoastSnap cradles (i.e., metal phone holders) installed at strategic coastal viewpoints, by placing their smartphones into the cradle to capture photos of the coast. Residents can also use the app's do-it-yourself adaptation to photograph the coastline without a designated phone cradle. By capturing daily or bi-weekly shoreline conditions through community participation, this method could provide valuable data that could fill

in the gaps of a purely satellite derived coastline monitoring program and allow for an even finer scale understand of coastline movement and risk in future reports.

Until such stations are installed, areas that are best for residents to upload do-it-yourself adaptation photos for the app have been highlighted (Figure 13.1). This setup would enable consistent visual tracking of shoreline changes along the accreting and erosion zones near the groynes.

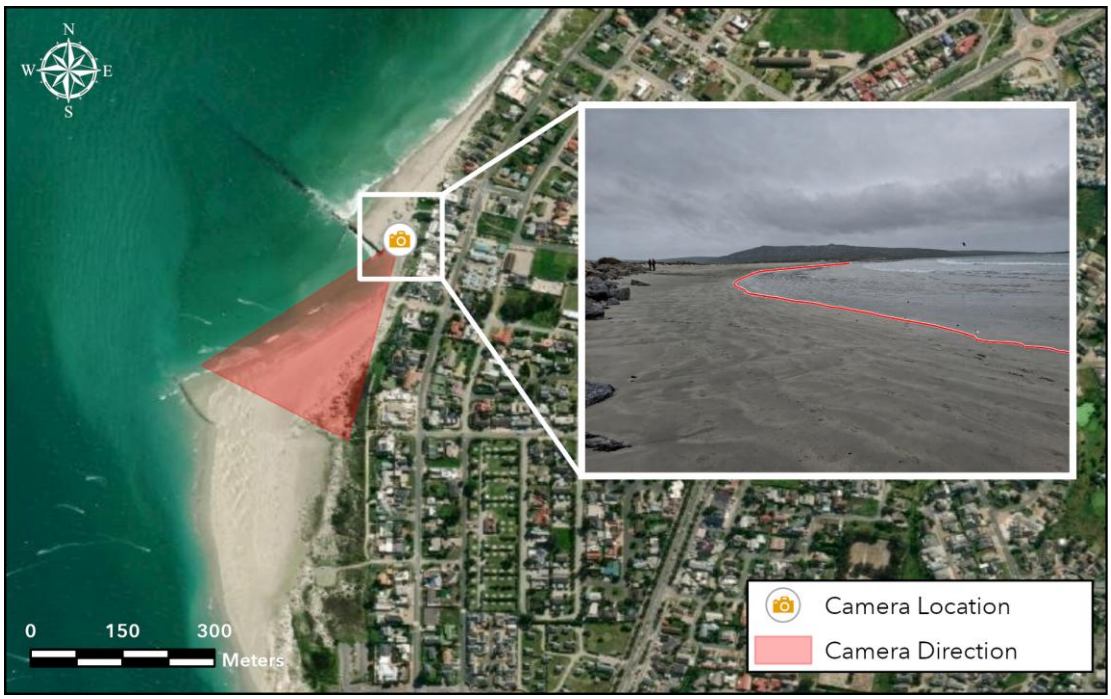


Figure 13.1. CoastSnap do-it-yourself camera location and viewing direction at Langebaan Beach. The inset photograph shows the visible water-land boundary, delineated by the red line.

The remote-sensing-based coastal monitoring programme for Saldanha Bay and Langebaan Lagoon should be continued and refined through the targeted collection of data needed to assess coastal vulnerability and identify the environmental drivers of erosion hotspots. Future monitoring should prioritise assessments of vulnerable areas, supported by in situ wave and atmospheric measurements and higher-resolution shoreline-change data. This approach will improve understanding of the processes driving erosion and accretion, strengthen mitigation strategies and support targeted coastal protection measures.

13.6 BENTHIC SYSTEMS

Monitoring of benthic infaunal communities from 1999–2026 has revealed patterns of decreased abundance and biomass following anthropogenic events, followed often by recovery. Recent findings indicate a marked improvement in these indices, suggesting some recovery in benthic infaunal community structure, though further monitoring is required to determine whether full recovery has been achieved.

Notable improvements include the return of the sea pen *Virgularia schultzei* to Big Bay and Langebaan Lagoon since 2004, as well as an increase in the biomass percentage of large, long-lived species such as the tongue worm *Listriolobus capensis* and several gastropods. Certain areas of Small Bay, which experience reduced water circulation leading to the accumulation of fine sediment, organic material, and trace metals (aggravated by anthropogenic inputs), still have impoverished infaunal communities. To ensure the continued improvement in the health of Small Bay's marine environment, it is recommended that stringent controls be placed on the discharge of effluents into Small Bay to facilitate the recovery of benthic communities and overall ecosystem health. The regularity (annually) and intensity of benthic infauna monitoring should continue. Continued baseline monitoring of the Elandsfontein macrobenthic community is likewise recommended to capture natural variability and provide a robust basis for detecting potential future impacts associated with the proposed phosphate mine, particularly those affecting groundwater quality and flows to Langebaan Lagoon.

The Big Bay reef habitat has been confirmed as an ecologically important habitat within the Saldanha Bay ADZ, including areas within the finfish aquaculture precinct. Initial surveys identified diverse epifaunal communities associated with both low-profile and more substantial reef outcrops. Subsequent bathymetric mapping in 2024 has improved understanding of the extent and distribution of the reef habitat, while the 2025 repeat ecological survey provided a more quantitative assessment of its associated epifaunal communities. Despite the absence of clear aquaculture-related effects, the ecological importance and spatial extent of the reef warrant continued monitoring. The reef provides habitat for diverse epifaunal communities and is considered important as nursery habitat for juvenile West Coast rock lobster. Given that a substantial proportion of the mapped reef occurs within the ADZ Big Bay precinct, particularly within the finfish area, future finfish aquaculture development should preferentially be directed towards the Outer Bay to minimise potential interactions with reef habitat. Continued standardised monitoring of established impact, control and baseline sites is recommended to detect any longer-term changes in community composition and to distinguish potential aquaculture-related effects from natural spatial variability.

13.7 FISH AND ECONOMICALLY IMPORTANT INVERTEBRATES

The results of the 2026 seine net surveys reaffirm the importance of Saldanha Bay and Langebaan Lagoon as critical nursery areas for juvenile fish species and economically important reef-associated invertebrates. They also highlight concerning declines in several priority fishery species, demonstrating the continued need for long-term monitoring and stronger management interventions.

The annual SOB seine net surveys remain a strong component of the monitoring programme, providing a unique long-term dataset on juvenile fish recruitment, diversity, and nursery habitat use. These surveys serve as an early warning system for stock collapse of key fishery species, most notably for white stumpnose, and should therefore be continued without interruption. Newer complementary techniques such as BRUVs (implemented by SANParks) and eDNA also add important information on cryptic or low-abundance species, while the introduction of biennial diving surveys of abalone and rock lobster has filled a longstanding gap in monitoring reef-associated invertebrates. Together, these methods provide a more complete picture of ecosystem health, and their continuation and expansion are strongly recommended. Improved monitoring of

recreational fishing effort and catches is particularly important, given the dominance of this sector in targeting species such as white stumpnose and elf, yet the absence of systematic catch-per-unit-effort data.

Management interventions are urgently needed to address the decline of several overexploited stocks. For white stumpnose, which remains at historically low abundance, a reduction in the daily bag limit from 10 to 5 fish per person, and an increase in the minimum size limit from 25 cm to 29–30 cm Tail Length, should be implemented as soon as possible. These measures have been recommended for more than a decade and would significantly improve stock recovery prospects. Elf, which shows evidence of both resident and migratory components within Saldanha Bay, has experienced poor recruitment since 2018. Given that this may represent a distinct stock, localised management and continued close monitoring are required.

The harder stock is heavily depleted, with biomass estimated at less than a quarter of pristine levels. Implementation of Horton et al. (2019)'s recommendations, which include reduced fishing mortality, increased gillnet mesh sizes, and stricter enforcement against illegal netting is necessary to facilitate recovery. Blacktail, which are highly resident in Small Bay, have shown almost complete absence of juveniles since 2008, and protecting the nursery habitats of Small Bay is therefore critical for their persistence. The similar scarcity of bluntnose guitarshark may also be cause for concern and warrants close future attention.

The Langebaan Lagoon MPA continues to play a key role in protecting fishery resources. Studies have repeatedly demonstrated that white stumpnose, elf, harders, and smooth hound sharks benefit from the MPA's protection. Pressures to weaken these protections by expanding access for gillnet fishers should be resisted. Instead, management efforts should focus on improving compliance and enforcement, ensuring that the benefits of the MPA continue to support stock resilience.

Finally, the socio-economic importance of recreational and small-scale fisheries in the region should not be underestimated. Future management decisions would be strengthened by integrating economic assessments of these fisheries, thereby capturing the true value of healthy fish and invertebrate populations to local communities and the regional economy.

13.8 BIRDS AND SEALS

Together with the five islands within the Bay and Vondeling Island slightly to the south, Saldanha Bay and Langebaan Lagoon provide extensive and varied habitat for waterbirds.

The islands of Malgas, Marcus, Jutten, Schaapen and Vondeling support nationally important populations of Cape Gannet, Swift Tern, Kelp and Hartlaub's Gull, four species of marine cormorant (Cape, Crowned, Bank and Whitebreasted), important populations of the endemic African Oystercatcher. Nationally important populations of African Penguin and Bank Cormorants also used to breed on the Saldanha islands, but the number of nesting pairs of these species has dwindled. 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 by DFFE (branch Oceans and Coasts), and bi-annually (summer and winter) in Langebaan Lagoon by citizen scientists as part of

the Coordinated Waterbird Counts (CWAC) programme started by the Animal Demography Unit of the University of Cape Town.

Declines in the numbers of seabirds breeding on the Saldanha Bay islands over the last two and a half decades can be attributed to several causes. In addition, the recent (July 2026) outbreak of Pilchard Herpes Virus (PHV) that has spread rapidly throughout the Namibian and South African pilchard stock could also have significant negative impacts on the upcoming 2026 and future seabird breeding success. Gannet and penguin diets shifted after the Australian PHV outbreak and declines in breeding success was recorded in the breeding seasons after the outbreak. A similar response in seabirds should be anticipated with the South African PHV outbreak.

Other factors, including predation by Cape fur seals, Pelicans and Kelp Gulls, are also contributing to current population declines for many seabird species on the islands in Saldanha Bay. Direct amelioration actions to decrease these impacts at the islands (Pelican Watch, problem seal culling) have had mixed results, with the former proving more effective than the latter. Cape fur seal and Kelp Gull predation continue to pose a major threat to seabird survival at the Saldanha Bay Island colonies. Increasing volumes of shipping traffic and associated underwater noise pose a further threat as it may negatively impact the feeding behaviour and hence survival of some sea bird species.

Decreasing numbers of migrant waders utilising Langebaan Lagoon over the last 40 years reflects a global trend. Resident wader numbers in the winter of 2022 had dropped to the lowest recorded in the 40-year count record with just 142 recorded. However, the most recent count of 1020 birds in the winter of 2025 suggests conditions may have improved. Drastic species level population declines in four migratory waders, including the Ruddy Turnstone, Red Knot, Grey Plover and Curlew Sandpiper have signified a downward trend in summer migratory bird numbers. Resident wader numbers appear to have stabilised since 2010 at around half of the pre-1990 levels, suggesting that management measures such as zonation have been effective in halting the declines. It is highly recommended that the status of coastal and wading bird species continues to be monitored and that these data are used to inform and assesses the efficacy of management interventions aimed at supporting recovery of the region's birds.

Cape fur seals are amongst the largest marine top predators found in and around Saldanha Bay. They are opportunistic, generalist feeders that have been shown to benefit from human activities including utilisation of discards from fishing boats, or taking fish directly from fisherman. In addition, seals compete with seabirds, such as penguins and gannets, as well as with commercial fisheries, for small pelagic fish which form a key part of their diets. Cape Fur Seals are also attracted to the aquaculture infrastructure within Saldanha Bay, in particular the rafts that they use extensively as haul outs for resting. Even though the overall seal population within Southern Africa has remained stable over the last 30 years, changes in population distribution in response to environmental and ecological drivers is ongoing. The recent discovery of rabies, that is likely endemic in the Cape fur seal population, requires vigilance and a rapid medical response should any person or their pet be bitten by a seal.

13.9 ALIEN AND INVASIVE SPECIES

Human induced biological invasions have become a major cause for concern worldwide. Most of the introduced species in South Africa have been recorded in sheltered areas such as harbours and are believed to have been introduced through shipping activities, largely via ballast water and hull-fouling. There is an increased need to implement more efficient protocols to prevent the introduction, and provide sufficient monitoring, of alien species.

Given ongoing discoveries and evolving species statuses, the list of alien species in South Africa is dynamic. The increasing discovery of new alien species in Saldanha Bay highlights the need for proactive management. As managing these species is a challenging, and resource-intensive, task prioritising actions is crucial. The primary focus should be on managing existing invasive species in Saldanha Bay, followed by measures to prevent future invasions. Surveys must continue to verify the presence of unconfirmed listed species through morphological taxonomic and genetic research. Additionally, dedicated sampling efforts are necessary to understand the abundance, distribution, and potential impacts of these alien species. Some species may require control or eradication programs for effective management. Furthermore, prioritising certain alien species for management is recommended, using impact-based ranking systems like the Environmental Impact Classification for Alien Taxa (Blackburn et al. 2014). However, its application in the marine context is limited due to the lack of species-specific knowledge. In South Africa, for example, impacts have been quantified for only 16% of taxa known to be alien (Alexander et al. 2015, Robinson et al. 2016). Studies investigating the impacts of these species in Saldanha Bay are needed to prioritise management actions.

Preventing further invasions should also be a priority, such as employing watchlists based on invasion history, pathways, and the biological traits of species. Routine monitoring, coupled with watchlists, enhances early detection and successful eradication. Although resource intensive, there has been recent success in South Africa using curated databases and citizen science data (iNaturalist) as a supplementary tool for alien species monitoring (Ackland & Robinson 2025, Ackland et al. 2025b). These platforms provide data that cover broad temporal and spatial scales, while not requiring funding.

Preventing further introductions also involves identifying and managing key pathways, with shipping activities identified as a significant contributor to marine introductions in South Africa. Approximately 95% of marine introductions are associated with shipping, emphasising the need for robust port control protocols (Robinson et al. 2020). Saldanha Bay, being a crucial international port with substantial shipping traffic, requires enhanced measures to monitor vessels entering its harbours. This includes the treatment of hull fouling and ballast water before port entry, along with regular monitoring for alien species. Recreational yachts, often overlooked as vectors, pose a considerable risk for introducing alien species (Peters et al. 2019). International yachts frequenting Saldanha Bay increase this risk, and currently, there are no regulatory measures to manage alien species fouling on yachts entering South Africa. Encapsulation shows promise as a tool to minimise and manage fouling on vessels, and similar experiments are encouraged for Saldanha Bay (Keanly & Robinson 2020).

In summary, managing invasive species in Saldanha Bay necessitates a dual approach: addressing existing species, particularly those with known impacts, through dedicated sampling efforts, and proactively preventing further invasions.

13.10 AQUATIC MACROPHYTES

Overexploitation, physical modification of coastal systems, nutrient and sediment pollution, the introduction of invasive alien species, and global climate change have compounding negative effects on macrophyte habitats and therefore need to be carefully monitored and managed, particularly in Langebaan Lagoon. Without these critical habitats, their associated provisioning, regulating and cultural services would be lost.

Investment in their restoration and preservation will also support ongoing aquaculture developments in the Bay. Aquaculture systems stand to benefit from certain ecosystem services provided by macrophytes, such as water filtration, oxygenation, the retention of phosphorous and nitrogen, and the removal of suspended particulate matter (de los Santos et al. 2020).

13.10.1 SEAGRASS

The high level of sub-annual variability in the growth and decay of seagrass beds is attributed to several interconnected natural and anthropogenic influences. For example, increased siltation in the Lagoon decreases light availability and thereby limits seagrass growth, which hinders the sequestration and retention of sediment and nutrients. Consequently, water quality must be monitored continuously at core seagrass meadows throughout the year to attribute the sub-annual variability in their extent to any single or a combination of parameters. The key parameters to be considered are water depth, water temperature, salinity, dissolved oxygen, pH, turbidity/ light availability, phosphates, and nitrate content. Seagrass condition should also be monitored at core meadows, with relevant parameters including meadow size, percent coverage, canopy height, shoot density, epiphyte load and the presence and abundance of flowers, fruits, or seeds.

Sediment samples in and surrounding seagrass meadows collected during dedicated seasonal surveys serve to identify any sub-annual changes to the particle size distribution of the sediments surrounding the meadows. Linking these physico-chemical observations with changes in their extent as observed by satellite will form the basis of a health assessment for core seagrass habitats that propagate transient meadows. Once a high-resolution long term continuous dataset (12+ month) has been collected, a status mapping effort can begin in conjunction with predictive species distribution models that will identify vulnerable beds and meadows for targeted restoration and preservation efforts. Furthermore, an analysis of the differences in spectral signatures of seagrasses related to various morphological traits such as leaf length, blade width, shoot density, in addition to epiphytic cover is lacking and would greatly improve the accuracy of a satellite-derived mapping effort.

In addition, seasonal surveys should be conducted to validate the extent of seagrass beds as mapped by the supervised classification scheme. The presence of the invasive bryozoan, *Amathia verticillata*, should also be closely monitored as this species has been identified as a threat to *Z. capensis* within the Lagoon, as the bryozoans spread rapidly, form large colonies and compete with *Z. capensis* for sunlight and suitable habitat

(Williams 2007) (Figure 12.17). The first study that identified the invasive bryozoan as present in Langebaan found a greater presence of *A. verticillata* around the head of the lagoon, around the Geelbek area (Ackland et al. 2025a), especially during summer months. SANParks, in collaboration with researchers from the University Stellenbosch, have conducted seasonal surveys of the invasive bryozoan. These surveys should be repeated annually to provide a valuable record of changes over time in seagrass cover in relation to the extent of A. verticillata.

Little is known on the species composition or distribution of epiphytes associated with *Z. capensis* in Langebaan Lagoon. Seasonal surveys of Saldanha should include the opportunistic collection of a few *Z. capensis* leaf blades from different patches of seagrass around the lagoon, so that seagrass-associated epiphytes can be identified under a dissecting microscope in a laboratory. The distribution of these epiphytes can then be mapped and compared between sites to provide insights into factors driving high epiphyte loads. This may assist future remote monitoring efforts, as certain algal species, such as the *Ceramium* species identified in Figure 12.15, absorb different wavelengths of solar radiation compared to seagrasses. If finer resolution imagery were to become available in the future, the different spectral wavelengths absorbed by the seagrass and epiphytes could be used to distinguish between the two to quantify area coverage across the lagoon for ongoing monitoring.

RESTORATION POTENTIAL

The large-scale loss of seagrass meadows is a global issue which has been receiving more attention in recent years. Given the concern over the loss of valuable ecosystem services associated with seagrass meadows, a wave of international restoration targets and policies (e.g., UN Strategic Plan for Biodiversity; Convention on Biological Diversity (CBD), 2014) have initiated global ecosystem restoration action, such as the UN Decade on Ecosystem Restoration (2021–2030). As substantial areas of seagrass meadows have been lost within the Langebaan Lagoon, several projects have investigated the potential of restoration projects (Watson et al. 2023). Admittedly, seagrasses are a very challenging and expensive ecosystem to restore, not only globally but particularly within a South African context. Challenges to seagrass restoration include dynamic and stressful coastal environments, the removal of stressors which are likely to have caused the loss of meadows (such as poor water quality from pollution or dredging) and the logistical challenge of acquiring and moving a minimum critical mass of seagrass to increase survival probability (van Katwijk et al. 2016). South African populations are often genetically distinct and highly adapted to their local environmental conditions, meaning that sourcing donor material from the same estuary is critical to increase the success rates of restoration attempts (van Katwijk et al. 2016, Smit et al. 2023).

Furthermore, to implement effective management programmes, there is a need to develop predictive species distribution models. Habitat Suitability Models (HSMs) predict the species or habitat distribution based on an understanding of environmental variables that influence and determine their distribution. Building HSMs facilitates the identification of the most suitable recipient sites for protection and restoration efforts, potentially increasing the success of restoration, while offering a standardised and objective decision-making approach (Watson et al. in press). Management aided by HSMs provide options for decisionmakers, whose objective may be conservation or restoration, or a combination of these depending on the management objectives. In

areas where seagrass is localised, actions may include additional conservation measures to promote the persistence of meadows (Watson et al. in press). For regions where seagrass is currently absent, management actions improving environmental conditions may promote natural colonisation and increase success of restoration efforts (Watson et al. in press).

While restoring seagrass ecosystems is challenging, it may become necessary in Langebaan Lagoon if seagrass meadows continue to decline. Restoration trials have reported positive growth in restored transplants at Oesterwal with donor material from the Centre Banks population. Overall survival of transplants was ~58% with an increase of ~112% in seagrass cover (Watson et al. 2023). Although restoration may become necessary, protecting and conserving existing seagrass meadows remains the most effective management strategy, given the low success rates and logistical challenges associated with restoration (Orth et al. 2006, Waycott et al. 2009).

13.10.2 Salt marsh

Salt marshes in Langebaan Lagoon have proved to be more stable, yet the fringe growth on the edges of their core habitats do exhibit sub-annual and seasonal variability in growth and decay. Fortunately, the designation of the Lagoon as a MPA and a Ramsar site has alleviated historic agricultural grazing pressures. As long as these designations remain, the primary threats to salt marsh are climate induced change; in particular, sea level rise. It is therefore imperative that climate forecasting and sea level rise risk analyses be conducted in future monitoring efforts in conjunction with shoreline analyses.

If salt marshes were to undergo a significant decline in the future, management and restoration actions such as the reducing of impacts (improving tidal exchange, ensuring freshwater inflow, preening encroaching development and reducing trampling), remediation, rehabilitation and restoration would likely be necessary and should be implemented according to the national framework presented by Adams et al. (2021). Monitoring programmes should assess the state of salt marshes by measuring indicators of physical condition, species composition, structural diversity, and ecosystem functioning.

13.10.3 Common reed

While it has been established from a groundwater assessment in 2014 that the phosphate mining operations by Kropz Elandsfontein are highly unlikely to have any impact on the groundwater quality and flow (see Chapter 5: Groundwater), Kropz Elandsfontein have opted to take a precautionary approach and carefully monitor any potential impacts on Langebaan Lagoon (Conrad 2014). Langebaan Lagoon is not fed by overland streams or rivers and it has been suggested that groundwater plays a significant role in sustaining the marsh ecosystems at the head of the lagoon (Valiela et al. 1990, Burnett & Stickle 2001). Reduced freshwater inflow into Langebaan Lagoon could result in the development of more extreme hypersaline conditions in the upper lagoon, killing flora and fauna sensitive to salinities in excess of normal seawater.

As part of the monitoring for potential impacts on the Lagoon from the ground water extraction, an AquaTroll 200 has been deployed for long-term tracking of water quality data in the area near Geelbek. While this is primarily to monitor estuary conditions in

relation to groundwater extraction, the data may in future also be used to correlate expansion and contraction of nearby *P. australis* beds with salinity and temperature regimes, as salinity is a key factor influencing distribution (Karberg 2015). This could be used to provide insights into distinguishing between natural vs anthropogenic impacts stressors that may limit the spread of new growth.

In addition to monitoring of the change in the extent of *P. australis*, monitoring the salinity and pH of the sediment and water contained directly within their populations at Geelbek at seasonal intervals is recommended. While salinity and depth to the water table are the predominant factors controlling their distribution, these additional parameters have a direct impact on a populations ability to germinate. In the absence of these additional data, it is not possible to differentiate between natural and anthropogenic stressors that limit the spread of new growth.

13.10.4 Citizen Science: Opportunities to get involved

The present approach to monitoring by remote sensing is effective at identifying core, stable habitats and general presence and absence trends in class cover. While the use of open-source imagery has the benefit of being cost-effective, the issue remains that a classification effort using 10 m resolution imagery cannot detect macrophyte patches smaller than 10 m^2 in size. This is further exacerbated by the fact that these coastal and marine habitats have a high level of interspecific competition, where many different species of vegetation grow in close proximity to one another. The combination of these limiting and confounding factors leads to a systemic underestimation of class extent. At minimum, it is recommended that targeted ground-truthing be conducted for seagrasses at Centre Banks and salt marshes at Geelbek, so that a species composition (and associated spectral) analysis can be conducted. The results will allow for a more accurate analysis of present extent and change over time.

In addition to opportunistic ground-truthing by the Anchor team, locals and visitors are encouraged to get involved in citizen science to assist in ground-truthing seagrass extent by using freely available apps such as iNaturalist (www.inaturalist.org, Figure 13.2) and Seagrass Spotter (seagrassspotter.org, Figure 13.3). For both, iNaturalist and Seagrass Spotter, users can upload photos of seagrass and associated epiphytes (or any other species) to global databases. In App algorithms aid in identification of species and the Apps are easy to use and accessible to everyone. They are fun and a great way to get children interested in the natural environment.

Both Apps aid in ground-truthing processes, particularly, in this case, for seagrass in Langebaan, by storing GPS locations and date information which can then be accessed by remote researchers. This provides information on transient patches of seagrass, as seagrass presence can be mapped in time and space across the lagoon and compared to results generated by remote sensing. The images are also useful in capturing on the ground information and, while they cannot be used for measurements, they do give an indication of leaf length and shoot density, as well as epiphyte load.

The more observations that are recorded on the apps by individuals, the bigger and better the available dataset will be and the more useful it is to researchers and can then be used to validate remote sensing results. This information is incredibly useful and the more community involvement, the more robust the dataset is. This is one example of how value

citizen science is, as it provides a much greater coverage and time log of species presence/absence data than what be collected by just researchers.

In addition to seagrass and associated epiphytes, citizen science on iNaturalist will also be a valuable tool for monitoring and identifying other macrophyte groups, such as salt marsh and reed species, as well as the spread of the alien bryozoan *A. verticillata* over time and across different seasons within Langebaan lagoon.

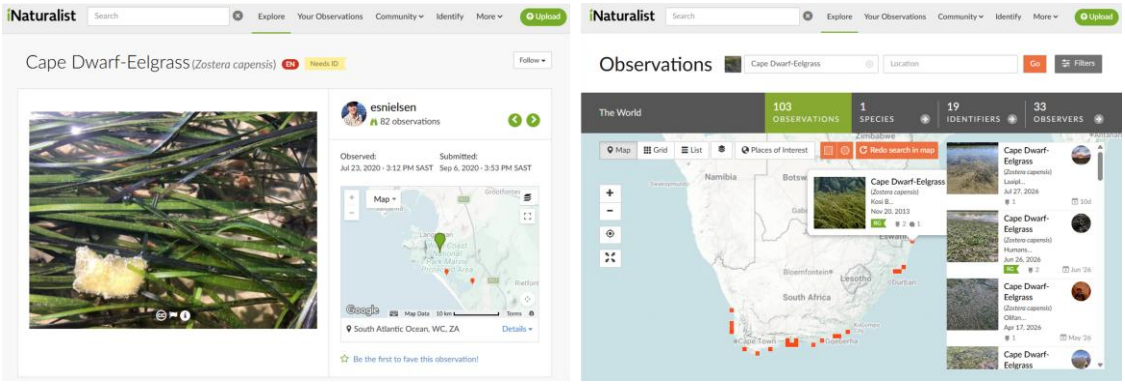


Figure 13.2. INaturalist webapp interface showing (left) an observation of Cape dwarf eelgrass in Langebaan lagoon with the uploaded image and GPS location; and (right) a map of Cape dwarf eelgrass observations made around the coast of South Africa.

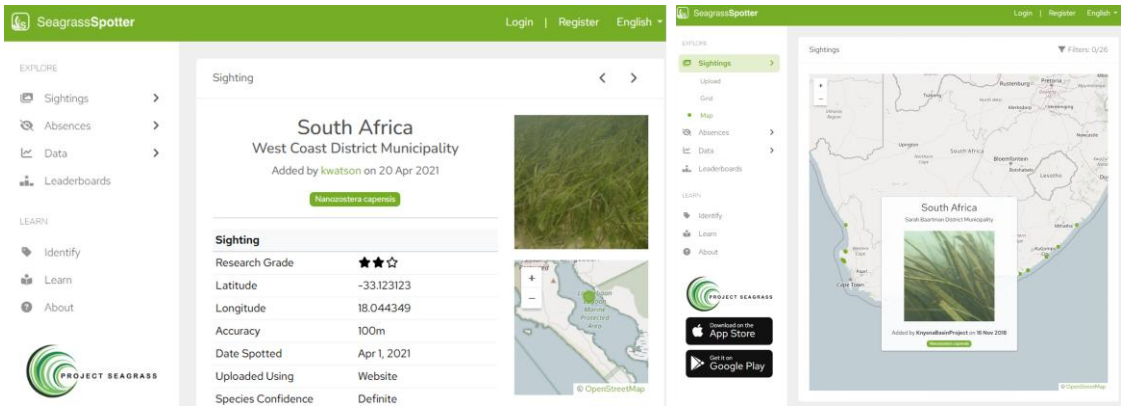


Figure 13.3. SeagrassSpotter webapp interface showing (left) an observation of Cape dwarf eelgrass in Langebaan lagoon with the uploaded image and GPS location; and (right) a map of Cape dwarf eelgrass sightings made around the coast of South Africa.

13.11 Summary

In summary, the environmental monitoring currently implemented in Saldanha Bay and Langebaan Lagoon (e.g. sediment, benthic macrofauna, birds, rocky intertidal communities, 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. Summary of environmental parameters reported on in the State of the Bay: Saldanha Bay and Langebaan Lagoon in 2026

Parameter monitored	Time period	Anthropogenic induced impact	Rating
ACTIVITIES AND DISCHARGES			
Wastewater treatment works (WWTW) discharges	Current–2026	Monitoring results show improvement in compliance of some of the parameters tested for both the Langebaan and Saldanha Bay WWTW, but also some declines in compliance. The surplus volume of treated effluent from the Saldanha WWTW, not used by local industry for irrigation or dust suppression etc., that enters the Bay via the Bok estuary is not currently known. It is therefore strongly recommended that a flow gauge be installed and monitored to record continuous discharge volumes entering the Bay. Where there is evidence of a deterioration in a water quality parameter over time, investigations should be made to ascertain and rectify the cause.	
Ballast water discharge volumes	Current–2026	The reporting style has changes to be more sensible and it seems like stricter rules are being enforced with regards to ballast water discharges as total discharges has been decreasing over the past few years which is a promising trend.	
GROUNDWATER			
Groundwater levels	2017–2026	Long-term monitoring indicates that groundwater levels across the borehole network remain stable, with no negative regional trends. This suggests that current wellfield abstraction and other groundwater uses are not imposing significant stress on the aquifer system. Key boreholes at Geelbek remain critical for detecting any potential impacts on freshwater flows to sensitive ecological areas such as Langebaan Lagoon, and this year's dedicated cross-section assessment of the Elandsfontein mine confirms that dewatering-related drawdown stays localised around the pit, with no evidence of it reaching the Lagoon. Continued monitoring is strongly recommended to ensure early detection of any potential future changes.	
Groundwater abstraction	2017–2026	Daily monitoring confirms that abstraction at both wellfields remains well within the licensed volumes authorised by DWS, with only 8.84% of Langebaan Road and 2.20% of Hopefield wellfield volumes abstracted by mid-2026. Volumes were notably lower than in previous years, largely due to extensive vandalism at both wellfields in September 2025. Continued monitoring and reporting of abstraction are recommended, together with timely repair of the affected infrastructure to restore normal operations.	
General groundwater quality (electrical conductivity; EC)	2017–2026	Monitoring of electrical conductivity shows generally stable water quality across the network. The upper aquifer exhibits higher conductivities due to natural mineralisation, while the lower aquifer, which hosts the municipal wellfield boreholes, shows lower values (better quality), except for localised variations at borehole G46092. EC trends along the Elandsfontein cross-section similarly show no consistent pattern linking the mine to the Lagoon. Ongoing EC monitoring is recommended to maintain a good understanding of aquifer health.	
WATER QUALITY			
Physical variables (temperature, salinity, dissolved oxygen (DO), nutrients and chlorophyll)	1974–2000, 2014–2025	DO levels in bottom water in Small Bay are frequently lower than they were historically or at least prior to port development. DO concentration is lower in Small Bay than in Big Bay and measurements below bivalve farms are consistently lower than at control sites. This is attributed to organic loading in Small Bay and reduced flushing time. No consistent changes are evident with any other physico-chemical parameters.	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
Current circulation patterns, current strengths, wave energy	1975 vs. 2020, 1960–2022	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., Iron Ore Terminal). Increasing significant wave heights near to Saldanha Bay over time (1960 to present day), supporting well-documented observations of changing weather and wave climates around Southern Africa.	
Microbiology	1999–2026	Counts of *E. coli* were largely acceptable in 2025 (19/20 sites = 'Excellent'). Generally, water quality is much higher than compared to historical data. Hoedjiesbaai recreational waters were rated as Excellent for the second year in a row. However, faecal coliform counts in nine of 10 Small Bay sites did not meet new stringent WQGs for mariculture use and although there have been improvements at some sites, others remain a concern (Hoedjiesbaai, Bok River Mouth). Historically, Big Bay has generally complied with mariculture water quality guidelines due to effective tidal flushing. In 2026, however, several sites showed a broader decline in microbiological water quality, characterised by persistently, but only slightly, elevated faecal coliform concentrations rather than isolated pollution spikes. 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, 2014–2026	Concentrations of lead in mussel flesh collected in research samples has improved since 2022 with all levels below the guideline limits. However, trace concentrations arsenic levels are now of concern considering revised guideline limits and recent exceedances thereof. Although elevated manganese levels have not been shown to have an adverse effect on marine life measures should be put in place to prevent excessive amounts of manganese dust from entering the Bay.	
COASTAL EROSION			
Transects along Leentjiesklip, Calypso and Mykonos beaches	2020–2026	Generally stable to slightly accreting, with mostly weak or non-significant wave correlations and localised erosion in some northern sections.	
Langebaan North Beach, from the groynes to the end of the rock revetment	2020–2026	Moderate seasonal variability, with higher winter waves linked to erosion despite stabilisation by coastal structures.	
Langebaan Beach between groyne 1 near Sleigh Street and groyne 2 near Noord Street	2020–2026	Persistent erosion of approximately 2.3 m/year, associated with higher south-westerly winter waves and restricted sediment supply between the groynes.	
Langebaan South Beach, between the groynes and the entrance to the lagoon	2020–2026	Stabilised and accreted by approximately 15.8 m/year near the groynes, with summer erosion occurring farther into the lagoon.	
Paradise Beach near the IOT	2020–2026	Stable to accreting (0.4–1.6 m/year), with no significant wave correlation, likely due to sheltering by surrounding infrastructure.	
Central Paradise Beach	2020–2026	Persistent erosion of 0.04–0.62 m/year, strongly associated with higher south-westerly winter waves and degradation of the beach, dunes and exposed calcrete.	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
Paradise Beach near the gabion walls outside Zeus Street	2020–2026	Stable to accreting by up to 1.0 m/year, with no significant wave correlation possibly due to local sheltering.	
BENTHIC SYSTEMS			
Species abundance, biomass, and diversity of benthic infauna	1999–2026	Benthic infaunal communities remain variable among sites, with abundance and biomass showing recovery and fluctuations following major dredging events. At Small Bay and Big Bay, the observed variability in abundance and biomass remained within the established range of temporal variation, with no evidence of a sustained change beyond historical levels. Spatial differences in community composition persist, particularly between Small Bay and Big Bay, and localised reduced diversity indicate that anthropogenic pressures continue to influence benthic communities.	
FISH			
Community composition and abundance of juvenile surf zone fish	1994–2026	Fish communities showed strong interannual variability, with diversity and abundance in 2026 remaining below average in Small Bay, although Big Bay diversity and abundance improved. White stumpnose and blacktail remain severely depleted, elf show some recent improvement, and harders remain overexploited despite high 2026 catches. Bluntnose guitarshark, which was absent in 2025 for the first time in the survey record, remained scarce in 2026 with only a single individual recorded.	
BIRDS			
Population numbers of key species in Saldanha Bay and islands	1977–2025	Populations of some 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 due to availability of small pelagic fish stocks; (2) predation of eggs, young and fledglings by other species; (3) starvation; (4) competition for food with the small pelagic fisheries; and, (5) collapse of the West Coast rock lobster stock upon which Bank Cormorants feed. The extended 20 km radius closure to small pelagic fishing around Dassen Island does not appear to have yet benefitted African Penguins nesting on the Saldanha Bay islands although Cape gannet numbers have increased slightly, possibly in response to small increases in pilchard biomass along the west coast. The recent detection of pilchard herpes virus in the South African stock is a serious concern with potential negative trophic impacts on sea birds reliant on this prey species.	
Population numbers of key species in Langebaan Lagoon	1976–2026	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. The apparent recovery in migratory wader numbers in Langebaan lagoon recorded during 2021 that was possibly due to reduced anthropogenic impacts during the COVID-19 pandemic, was short-lived and subsequent counts were again well below historical levels. Numbers of resident waders have also declined and is likely due to changes in the lagoon itself, particularly increased disturbance at historically important, feeding sandflats.	
SEALS			
Cape fur seal population numbers	1970–2025	Cape fur seal population stable over the past 30 years.	
ALIEN AND INVASIVE SPECIES			
Total number of alien and invasive species	2025–2026	Forty-seven alien species are now confirmed from Saldanha Bay and/or Langebaan Lagoon, of which 40 are considered invasive, and two naturalised. No new alien species have been reported since 2025. The highest number of individuals of the Western pea crab	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
in Saldanha Bay and Langebaan Lagoon		*Rathbunixa occidentalis* was recorded since monitoring began in 2004. The previously recorded alien anemone *Cylista ornata* has recently been correctly identified as *Cereus pedunculatus*. Preliminary genetic results from the Joint Institute for Nuclear Research (Russia) confirm the presence of *Mytilus edulis*, or a hybrid of this species and *M. galloprovincialis*, in Saldanha Bay.	
AQUATIC MACROPHYTES			
Seagrasses	2015–2026	Direct and indirect anthropogenic impacts such as increased siltation resultant of dredging activities, physical disturbance (boat activity and trampling), interspecific competition with alien invasives (such as *Amathia verticillata*) introduced via fouling on hulls of visiting vessels, and the effects of pollution (specifically eutrophication) all impact the health and growth of seagrasses and reduce their cover in the Lagoon.	
Salt marsh	2015–2026	Historic agricultural (grazing) stressors have been alleviated since the implementation of the MPA and protection under SANParks. Coastal development is the primary concern as urban areas rapidly expand.	
Common reed	2015–2026	No discernible impacts on common reed extent as a result of groundwater extraction by Kropz Elandsfontein phosphate mine.	

13.12 CLOSING STATEMENT

The 2026 State of the Bay report adds another important year of data to the long-term record of environmental change in Saldanha Bay and Langebaan Lagoon. Almost three decades of monitoring have demonstrated the value of consistent, long-term datasets in a system characterised by substantial natural variability and continued industrial, urban, recreational and conservation pressures. The programme provides an increasingly robust basis for distinguishing natural variability from longer-term environmental change and for identifying where management intervention may be required.

The 2026 results present a mixed but generally stable picture of the system. Several indicators show encouraging signs of improvement. However, important pressures remain. Small Bay continues to be particularly sensitive to restricted circulation, organic enrichment and contaminant accumulation, while elevated trace-metal concentrations in shellfish and continued concerns regarding several fishery species highlight ongoing environmental and resource-management challenges. Severe declines in some seabird populations, together with emerging pressures such as disease, invasive species and changing oceanographic conditions, further demonstrate the vulnerability of components of the system.

The findings also highlight the difficulty of attributing environmental change to individual pressures. Climate change and changing oceanographic conditions are likely to become increasingly important drivers of future change, alongside continued development, shipping, aquaculture, fishing and other human activities. This reinforces the need to consider cumulative pressures and the broader environmental context when planning and assessing future development.

The continued development of Saldanha Bay as an industrial and port centre makes long-term environmental monitoring increasingly important. Maintaining the core monitoring datasets, while incorporating complementary approaches such as remote sensing,

molecular techniques and citizen science, will improve the ability to detect emerging changes and assess the effectiveness of management interventions. Continued collaboration between government, industry, researchers, conservation agencies and other stakeholders will be essential to ensure that monitoring results are translated into effective adaptive management.

Improvements in some components demonstrate that effective management can deliver environmental benefits, while continued declines or vulnerabilities in others show that these gains cannot be taken for granted. Continued long-term monitoring, supported by timely and coordinated management responses, will therefore remain essential to maintaining the ecological integrity and associated socioeconomic benefits of Saldanha Bay and Langebaan Lagoon. The 20th State of the Bay report will provide an important milestone in this ongoing record and an opportunity to reflect on three decades of environmental change and management in the system.

Saltmarsh near Geelbek

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GLOSSARY

Alga(e): An informal term for any organisms of a large and diverse group of photosynthetic organisms that are not land plants, and includes species from multiple distinct clades, ranging from unicellular microalgae, such as cyanobacteria, to multicellular macroalgae such as kelp.

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).

Amphipod: Any of a large order (Amphipoda) of small crustaceans with a laterally compressed body.

Anthropogenic: Relating to or resulting from the influence of human beings on nature.

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.

Ascidians: Sea squirts like redbait.

Benthic: Pertaining to the environment inhabited by organisms on or in the bottom of a body of water such as the ocean. Organisms living in this zone are collectively referred to as the "benthos".

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.

Biofouling: The undesirable accumulation of microorganisms, plants and/or animals on wet surfaces that have a mechanical function, causing structural or other functional deficiencies.

Biomass: The mass of living biological organisms in an area or ecosystem.

Biota/biotic: All the plant and animal life of a particular region.

Bivalves: A class of marine and freshwater molluscs that have laterally compressed bodies enclosed by a shell consisting of two hinged parts. Includes oysters, mussels, and scallops.

Brachiopod: Lamp shells, which have a chalky shell, distinguished by having dorsal (upper) and ventral (lower) shell-valves and a fleshy stalk. Valves are hinged at the rear end, while the front can be opened for feeding or closed for protection.

Bryozoan: A "moss animal". Build colonies of tiny individuals, each enclosed in a tiny skeleton and crowned filter feeding tentacles.

Cloud Masking: Process of identifying and excluding cloud-covered pixels in satellite imagery to enable accurate shoreline detection and analysis.

Cnidarians: A phylum that includes jellyfish, hydroids, sea anemone and corals.

Coastline Hotspots: Localized areas where the shoreline exhibits significantly greater rates of movement—either erosion or accretion—compared to adjacent coastal sections over a given time period.

Coastal buffer zone: A natural, usually protected, strip of land separating the ocean or beach from human building infrastructure. Helps to prevent erosion and protect coastal fauna and flora.

Coliforms: Bacteria that are always present in the digestive tracts of warm-blooded animals.

Colony-forming unit (cfu): A unit which estimates the number of microbial cells in a sample that are viable or able to multiply via binary fission under the controlled conditions.

Community structure: The number of species in a community and the relative number of individuals per species.

Community: A group or association of populations of two or more different species occupying the same geographical area at the same time.

Conductivity: A measure of the ability of water to pass an electrical current. Because dissolved salts and other inorganic chemicals conduct electrical current, conductivity increases as salinity increases.

Constellation (regarding satellites): A coordinated group of satellites within the same mission that operate in complementary orbits to provide continuous and consistent Earth observation data.

Contaminants: Biological or chemical substances or entities, not normally present in a system, capable of producing an adverse effect in a biological system, seriously injuring structure or function.

Cosmopolitan: Occurs all over the world.

Critical Biodiversity Area (Natural): Areas in a natural/near-natural ecological condition. Together with MPAs, CBAs are required to meet biodiversity targets so that a representative sample of coastal and marine biodiversity can persist. Management objectives include maintaining in natural or near natural ecological condition.

Critical Biodiversity Area (Restore): Areas in a degraded or secondary condition and which should be restored. Management objectives include improving ecological condition, and, in the long-term, restoring to a natural or near-natural condition or as close to that as possible. Further deterioration in ecological condition must be avoided, and options for future restoration must be maintained. Only low-impact, biodiversity-sensitive land-uses are appropriate.

Crustaceans: Invertebrate animals that constitute one group of arthropods that are traditionally a part of the subphylum crustacea, a large, diverse group of mainly aquatic arthropods including the more familiar decapods (shrimps, prawns, crabs, lobsters and crayfish), seed shrimps, branchiopods, fish lice, krill, remipedes, isopods, barnacles, copepods, opossum shrimps, amphipods and mantis shrimp.

Cryptogenic: Species of unknown origin.

Cumulative impacts: Direct and indirect impacts that act together with current or future potential impacts of other activities or proposed activities in the area/region that affect the same resources and/or receptors.

Detritus: Waste or debris of any kind.

Diatoms: A large group comprising several genera of algae, specifically microalgae, found in the oceans, waterways and soils of the world, and which are unicellular organisms occurring either as solitary cells or in colonies. Their cell walls are made of silica.

Dissolved oxygen: A measure of the amount of oxygen dissolved in the water. The amount of oxygen available to living aquatic organisms.

Diversity: The number of different species and relative abundance of each of those species present in an ecosystem or in a community. Also see species diversity.

DNA barcoding (sequencing): A method of species identification using short segment of DNA from a specific gene. Every species has its own barcode. These DNA barcodes can be compared to a reference library to provide an ID (synonyms: metabarcoding).

Dredging: The act of removing silt and other material from the bottom of bodies of water.

Ebb-delta: A large underwater mound or accumulation of sand deposited at the seaward entrance of tidal inlets or estuaries.

Echinoderms: Invertebrate organisms with adults that are usually characterised by their radial symmetry (usually five-point) and incudes sea urchins, sea cucumbers, feather stars, and sea stars.

Ecological Support Areas (ESA): Areas not essential for meeting biodiversity targets, but that play an important role in supporting the functioning of Protected Areas and CBAs and are often vital for delivering ecosystem services. The management objective is to maintain these sites in a functional, near-natural state. Some habitat loss is acceptable.

Ecologically or Biologically Significant Marine Areas (EBSAs): Defined by the Convention on Biological Diversity (CBD) as "geographically or oceanographically discrete areas that provide important services to one or more species/populations of an ecosystem or to the ecosystem as a whole, compared to other surrounding areas or areas of similar ecological characteristics, or otherwise meet the [EBSA] criteria".

Ecosystem Engineer: Any species that creates, significantly modifies, maintains or destroys a habitat. These organisms can have a large impact on species richness and landscape-level heterogeneity of an area.

Ecosystem: A biological community of interacting organisms and their physical environment—a complex network or interconnected system.

Ecotone: A transitional area between two different ecosystems or communities, exhibiting characteristics of both and often containing unique species. It is a zone where ecological communities overlap and integrate, leading to higher biodiversity in some cases.

eDNA: Environmental DNA is organismal DNA that can be found in the environment. It originates from cellular material shed by organisms (via skin, excrement, etc.) into aquatic or terrestrial environments that can be sampled and monitored using new molecular methods.

El Niño: El Niño and La Niña are two opposing phases of the El Niño-Southern Oscillation (ENSO) cycle, a climate pattern that influences weather and climate globally. El Niño is often associated with below-average rainfall and drought conditions in South Africa, particularly during the summer months (December to February).

Endemic: Endemism is the state of a species being found only in a single defined geographic location, such as an island, state, nation, country or other defined zone

Environment: The external circumstances, conditions and objects that affect the existence of an individual, organism or group. These circumstances include biophysical, social, economic, historical and cultural aspects.

Environmental Authorisation: Permission granted by the competent authority for the applicant to undertake listed activities in terms of the NEMA EIA Regulations, 2014.

Environmental Impact Assessment: A process of evaluating the environmental and socio-economic consequences of a proposed course of action or project.

Epibiont: An organism that lives on the surface of another living organism.

Epifauna: Organisms living on or just above the bottom sediments of a body of water.

Faecal coliforms: One type of faecal indicator and refer to a group of bacteria commonly found in the faeces of humans and warm-blooded animals.

Fauna: General term for all the animals found in a particular location.

Filter-feeders: Animals that feed by straining suspended matter and food particles from water.

Flora: General term for all the plant life found in a particular location.

Fouling communities: Communities of organisms found on artificial surfaces like the sides of docks, marinas, harbours, and boats, essentially fouling these structures.

Functional group: A collection of organisms of specific morphological, physiological, and/or behavioural properties.

Gastropods: A large and diverse class of molluscs of the phylum Mollusca characterised by their distinctive spiral shells, though some species have reduced or absent shells. These organisms have a muscular foot located at the ventral side of their body that is used for movement.

Genus (singular)/ genera (plural): A taxonomic rank referring to a group of closely related species that share common ancestry and exhibit similar characteristics. Organisms within the same genus have more similarities than those in different genera but are still distinct species.

Government Gazette: An official Government publication used to publish acts and bills, regulations and notices in terms of acts, changes of names, company registrations etc.

Gravel: Sediment with a grain size ranging between 2 mm and 64 mm.

Grazer: A 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.

Guano: The excrement of seabirds and bats. Can be used as fertiliser.

Hybridisation: The mating of genetically differentiated individuals or groups, forming hybrid organisms genetically distinct from either parent.

Hydrodynamic forces: Referring to forces acting on or exerted by water.

Hydrogeology: The study of groundwater. Sometimes referred to as geohydrology or groundwater hydrology.

Hydroid: Pertaining to that form of hydrozoan that is asexual and grows into branching colonies by budding.

Hydrostatigraphic: The classification of geological units based on their water-bearing properties. Grouping rocks or sediments into aquifers, aquitards, or aquicludes according to their ability to store and transmit groundwater.

Image classification: The process of assigning land cover classes to image pixel values.

Indigenous: Species within the limits of their native range (synonyms: native).

Infauna: Organisms living within bottom sediments of a body of water, e.g., polychaetes, amphipods, isopods.

Intertidal: Also known as the littoral zone, is the area of shoreline that is exposed to the air during low tide and underwater during high tide. It is the region where land and sea meet, and it experiences regular fluctuations in water level and exposure to air.

Introduced species: Non-native species that have been intentionally or unintentionally brought into a region with the help of humans.

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: Large algae or seaweeds from the family Laminaria of the order Laminariales, the more massive brown algae.

Linear regression rate: An ordinary least square regression of shoreline change over time

Macrobenthos/macrofauna: Animals retained by a 1 mm-mesh sieve. Macrobenthic invertebrates are defined as organisms that live on or inside the deposit at the bottom of a water body.

Macrophyte: An aquatic plant large enough to be seen by the naked eye.

Marchins squares contouring: A method that traces the boundary between land and water pixels in satellite images to map the coastline.

Mariculture: A branch of aquaculture involving the cultivation of marine organisms.

Marine Protected Areas (MPA): Areas formally protected in terms of the Protected Areas Act. The Management objective is defined by the management plan of each MPA.

Membrane Filter counts (reported as colony forming units (cfu)): Involves isolating and counting discrete colonies of bacteria.

Mytilid mussels: Any bivalve mollusc of the order Mytilida.

National Biodiversity Assessment (NBA): The threat status and sensitivity of different habitat types based on biodiversity (richness, uniqueness, spatial extent of the habitat type) and exposure levels to natural disturbance or environmental perturbations.

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.

Organic matter: Matter composed of organic compounds that have come from the faeces and remains of organisms such as plants and animals.

Other Natural Areas: Areas not currently identified as a conservation priority but retain most of their natural character and perform a range of biodiversity and ecological infrastructure functions. The management objective includes minimising habitat and species loss and ensure ecosystem functionality through strategic landscape planning.

Otsu threshold: A method in image processing that automatically finds the best cutoff to separate the main objects from the background in an image.

Pan-sharpening: An image fusion technique that combines a high-resolution panchromatic image with a lower-resolution multispectral image to create a higher-resolution multispectral image.

Phylum (singular)/ phyla (plural): One of the major hierarchical levels used to group and categorise living organisms based on their shared characteristics and evolutionary relationships

Phytoplankton: Ocean dwelling microalgae that contain chlorophyll and require sunlight in order to live and grow.

Piezometer: A non-pumping borehole, generally of small diameter, for measuring the elevation of a water table or collecting water samples.

Polychaetes: Segmented marine worms commonly called bristle worms due to each segment bearing a collection of bristles.

Polychromatic: Having various or changing colours; multicoloured.

Porifera: Scientific name for sea sponges.

Probability density function: A function whose value at any given sample in the sample space can be interpreted as providing a relative likelihood that the value of the random variable would be equal to that sample.

Propagule pressure: The quality, quantity, and frequency of invading organisms.

Ramsar site: A wetland site designated to be of international importance under the Ramsar Convention, also known as "The Convention on Wetlands of International Importance", an international environmental treaty signed on 2 February 1971 in Ramsar, Iran, under the auspices of UNESCO.

Raster: A matrix of cells (or pixels) organized into rows and columns (i.e., a grid) where each cell contains a value representing information, such as land class. Rasters are digital aerial photographs, imagery from satellites, digital pictures, or even scanned maps.

Redox (Reduction Potential): The tendency of a chemical species to either be reduced by accepting electrons or oxidized by donating electrons.

Remote sensing: The science of acquiring information about an object or area from a distance, without physical contact. It involves using sensors, often on satellites or aircraft, to detect and measure energy reflected or emitted from the Earth's surface.

Scavenger: An animal that eats already dead or decaying animals.

Seabirds: Birds that are adapted to life within the marine environment.

Semigration: Short for semi-emigration, the act of moving to another location within a home country.

Sentinel-2x: European optical imaging satellite providing global coverage 10 m resolution imagery. 2A was launched in 2015, 2B in 2017, 2C in 2025, and 2D is planned for launch in 2028.

Shannon Diversity Index: Considers both species richness and evenness of the species' abundance, providing a single number that reflects the distribution of individuals among species in a community.

Species complex: A group of closely related organisms too similar in appearance to reliably distinguish based on morphological characteristics, e.g., mytilid species complex.

Species richness: The number of different species represented in an ecological community. It is simply a count of species and does not take into account the abundance of species.

Species: A category of biological classification ranking immediately below the genus, grouping related organisms. A species is identified by a two-part name; the name of the genus followed by a Latin or Latinised un-capitalised noun.

Spectral (profile) signature: The unique pattern of electromagnetic radiation (light) that a material or object reflects, absorbs, or emits at different wavelengths.

Subtidal: The marine habitat that lies below the level of mean low water for spring tides.

Supervised classification: The process of assigning class values to grid cells in a raster GIS database based on the spectral reflectance values of similar grid cells that have already been classified.

Supratidal: The coastal area above the high tide line that is regularly splashed by seawater but not submerged or inundated, except during very high tides or storms.

Suspension feeder: An aquatic animal which feeds on particles of organic matter suspended in the water, especially a filter feeder.

Taxon (singular)/ taxa (plural): Also referred to as a taxonomic group. Refers to any unit used in the science of biological classification, or taxonomy.

Thermocline: An abrupt temperature gradient in a body of water, marked by a layer, above and below which the water is at different temperatures.

Tides: The rise and fall of sea level, usually twice a day, due to gravitational forces by the moon. Spring tides have higher high tides and lower low tides, whereas neap tides have lower high tides and higher low tides.

Trace metals: Elements such as chromium, cobalt, copper, iron, magnesium, selenium, and zinc that normally occur at very low levels in the environment. High levels of these same metals can be toxic.

Training sample: Location information (either points or polygons) within or encompassing a known associated land cover class. An image classification algorithm uses the training samples, saved as a feature class, to identify the land cover classes in the entire image.

Transects: A survey line or path along the beach, typically running perpendicular from the high tide line to the low tide line or from the backshore to the foreshore.

Unsupervised classification: A type of machine learning algorithm that groups data points into clusters based on their inherent similarities without any prior knowledge or labelled training data.

Updrift side: The side "upstream" relative to the direction sediment naturally travels along the coast

Validation (remote sensing): The process of verifying the accuracy and reliability of remotely sensed data by comparing it with independent, high-accuracy reference data, often obtained through ground-based measurements or other reliable sources. It ensures that the data collected by sensors like satellites or aircraft accurately reflects real-world conditions and is suitable for intended applications.

Vectors: The specific means of introduction within a pathway. For example, if a species has been introduced via the shipping pathway, the vector of introduction can be hull-fouling or ballast water.

Vertebrates: Animals with bony or cartilaginous axial endoskeleton—known as the vertebral column, spine or backbone.

Waders: Birds of the order Charadriiformes commonly found wading along shorelines and mudflats in order to forage for food crawling or burrowing in the mud and sand, usually small arthropods such as aquatic insects or crustaceans.

Zooid: An animal arising from another by budding or division. Specifically, a single organism that is part of a colonial organism.

ABBREVIATIONS AND ACRONYMS

ADZ	Aquaculture Development Zone
AZE	Acceptable Zone of Effect
AI	Artificial Intelligence
AMBI	AZTI Marine Biotic Index
AZE	Acceptable Zone of Effect
BRUV	Baited Remote Underwater Video
BSP	Biodiversity Spatial Planning
CBA	Critical Biodiversity Area
CBD	Convention on Biological Diversity
cfu	Colony forming units
COD	Chemical Oxygen Demand
CSIR	Council for Scientific and Industrial Research
CVI	Coastal Vulnerability Index
CWAC	Coordinated Waterbird Counts
dbRDA	distance-based redundancy analysis
DEA	Department of Environmental Affairs
DEA&DP	Western Cape Department of Environmental Affairs & Development Planning
DFFE	Department of Forestry, Fisheries and the Environment (previously DEA and DAFF)
DistLM	Distance-based linear model
DNA	Deoxyribonucleic acid
DO	Dissolved Oxygen
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
EA	Environmental Authorisation
EBSA	Ecologically or Biologically Significant Marine Area
ECMWF-ERA	European Centre for Medium-Range Weather Forecasts-Reanalysis
eDNA	Environmental DNA
EIA	Environmental Impact Assessment
EMF	Environmental Management Framework
EMP	Estuary Management Plans
EMPr	Environmental Monitoring Programme
ERL	Effects Range Low
ESA	Ecological Support Area
GA	General Authorisation
GBK	Geelbek Boreholes
GDA	General Discharge Authorisation
GDP	Gross Domestic Product
GDPR	Gross Domestic Product by Region
GEOSS	GEOSS South Africa (Pty) Ltd
GG	Government Gazette
GN	Government Notice
HSM	Habitat Suitability Model
ICMA	National Environmental Management: Integrated Coastal Management Act (No. 24 of 2008)
IDP	Integrated Development Plan
IDZ	Industrial Development Zone

IOT	Iron Ore Terminal
IUCN	International Union for Conservation of Nature
LNG	Liquified Natural Gas
LPG	Liquid Petroleum Gas
LRR	Linear Regression Rate
mamsl	Metres above mean sea level
mbgl	Metres below ground level
MLRA	Marine Living Resources Act (No. 18 of 1998)
MNDWI	Modified Normalized Difference Water Index
MPA	Marine Protected Area
MPT	Multi-Purpose Terminal
MSP	Marine Spatial Planning
MW	megawatt
NBA	National Biodiversity Assessment
NEMA	National Environmental Management Act (No. 107 of 1998)
NEMBA	National Environmental Management: Biodiversity Act (No. 10 of 2004)
NEMPAA	National Environmental Management: Protected Areas Act (Act No. 57 of 2003)
nMDS	Non-metric Multidimensional Scaling
NOAA	National Oceanic and Atmospheric Administration
NWA	National Water Act (No. 36 of 1998)
PBH	Production Borehole
PDF	Probability Density Function
PHV	Pilchard Herpes Virus
ppt	Parts per thousand
PST	Paralytic shellfish toxins
PSU	Practical Salinity Units
RDAG	Red Dust Action Group
RO	Reverse Osmosis
SAET	Shoreline Analysis and Extraction Tool
SANBI	South African National Biodiversity Institute
SANDF	South African National Defence Force
SANParks	South African National Parks
SBM	Saldanha Bay Municipality
SDF	Spatial Development Framework
SEA	Strategic Environmental Assessment
SIMPER	Similarity Percentages
SNP	SANParks Boreholes
SOB	State of the Bay (State of Saldanha Bay and Langebaan Lagoon)
SWH	Significant Wave Height
SWIR	Shortwave infrared
TNPA	Transnet National Ports Authority
TOC	Total Organic Carbon
TON	Total Organic Nitrogen
tpa	Tonnes per annum
WCDM	West Coast District Municipality
WCNP	West Coast National Park
WQG	Water Quality Guidelines
WQT	Saldanha Bay Water Quality Trust (formerly SBWQFT: Saldanha Bay Water Quality Forum Trust)

APPENDICES

13.13 APPENDIX 1: LONG-TERM WATER LEVEL AND ELECTRICAL CONDUCTIVITY INFORMATION FOR THE SALDANHA BAY LOCAL MUNICIPALITY MONITORING BOREHOLES

Table 13.2. Details and field measurements of the Saldanha Bay Local Municipality Monitoring Boreholes. Source: GEOSS 2026.

Borehole	Coordinates		Elevation (mamsl)	Aquifer	Collar height (m)	Borehole depth (m)	Field Comment
	East	South					
G46064	18.23050	-33.00985	76	Lower Aquifer	0.28	83	Generally, fluctuations at this borehole are due to abstraction occurring at the Langebaan Road and Hopefield Wellfields. There are no concerns at this site, as groundwater levels do not fall below 35 mbgl. Manual EC measurements have increased over the past three monitoring periods, but remain well below the SANS241 drinking water limit. The EC values fluctuate between 50.3 mS/m and 65.4 mS/m.
G46025	18.238470	-33.012720	80	Upper Aquifer	0.63	65	A long-term dewatering trend was noted at this site from the start of monitoring until June 2023, when groundwater levels stabilised at around 14 mbgl. Manual EC measurements marginally increased over the past three monitoring periods, but remain low (within drinking standards). The EC values fluctuate between 48 mS/m and 62.1 mS/m.
G46031	18.179020	-33.024110	54	Unknown	0.61	88	A recovering trend in water levels has been observed since 2017. However, groundwater levels have remained relatively stable at approximately 8 mbgl since February 2024. EC varies between monitoring visits, with an overall decreasing trend (improvement) seen since the start of monitoring.
G46030	18.183600	-33.035270	66	Bedrock (shale)	0.60	119	The water level is stable over the long term at approximately 12 mbgl. There are no concerns at this site. EC measurements vary between monitoring visits with no observable trend.
BG00062	18.111760	-33.045360	23	Bedrock (granite)	0.52	27	Between 2015 and 2020, water levels at this site declined due to the regional drought. Over the past three years, however, water levels have risen in response to above-average rainfall in the area. The EC at this borehole is very high (exceeds the recommended SANS 241 drinking water guideline of 170 mS/m). This is, however, not of concern to the wellfield water, as this borehole is drilled into the bedrock aquifer, and the wellfield is drilled into the lower aquifer.
G33313	18.128970	-33.106630	63	Upper Aquifer	0.10	57	Groundwater levels remain relatively stable over the long term at approximately 11.5 mbgl. There have been no concerns noted at this site. EC measurements show natural variation and fluctuate between 105.2 mS/m and 156 mS/m with no observable trend.
G46092	18.087620	-32.944730	26	Lower Aquifer	1.05	28	This borehole remains artesian. Overall, ECs are variable with no observable trend. EC at this site remains above the SANS 241 guidelines (170 mS/m). There are no concerns as this

Borehole	Coordinates		Elevation (mamsl)	Aquifer	Collar height (m)	Borehole depth (m)	Field Comment
	East	South					
							borehole is not situated within the recharge flow path to the Langebaan Road Wellfield, and has historically been characterised by elevated EC.
G33327	18.127610	-32.963480	42	Unknown	0.4	83	An overall dewatering trend is observed since the start of monitoring; however, water levels have remained stable at approximately 16 mbgl since 2000. Fluctuations in water levels are likely due to abstraction taking place at the Langebaan Road Wellfield. EC at this site fluctuates between monitoring visits with no observable trend.
BG00054	18.157050	-32.961460	31	Unknown	0.30	45	There are no concerns at this site. A long-term rise is observed in water levels at this borehole. Downward fluctuations observed in the telemetry data are likely due to nearby farmers abstracting groundwater from the upper aquifer. EC measurements have increased over the last two monitoring site visits by 3 mS/m, but values remain within the expected range for this borehole.
G33246	18.336530	-32.901000	9	Upper Aquifer	0.10	25	Groundwater levels at this site fluctuate between 4 mbgl and 6 mbgl, with an overall rising trend observed over the long term. Fluctuations in EC are a result of natural variation, with no concerns noted.
G33326	18.271270	-32.933970	37	Upper Aquifer	0.10	80	Marginal fluctuations in water levels are observed at this borehole due to wellfield abstraction. An overall rising trend in water levels is observed over the long term. The EC has increased to 101.2 mS/m, but remains within the historical range for this site. No concerns are noted at this site.
G46065	18.271270	-32.933970	30	Lower Aquifer	0.44	34	Marginal fluctuations are observed in water levels at this borehole (2 mbgl and 4 mbgl), with an overall rising trend observed since the start of monitoring. Electrical conductivity values vary between monitoring events, with a decreasing (improvement) trend observed over the past year.
G46093	18.12070	-32.97735	52	Lower Aquifer	0.61	107	Groundwater levels exhibit an overall rising trend over the long term, suggesting healthy aquifer conditions, with good recovery and no evident adverse impacts from abstraction in the area. Electrical conductivity (EC) continues to fluctuate between monitoring events, with no clear long-term trend observed.
G46094	18.12074	-32.97730	52	Lower Aquifer	0.64	42.54	Similar to borehole G46093, a few metres away, groundwater levels have remained relatively stable since the start of monitoring, with minor fluctuations being attributed to natural variation. EC values continue to fluctuate between monitoring events, with no clear long-term trend observed.
G46089	18.09193	-33.03110	12	Upper Aquifer	0.10	9.20	A rising trend in water levels has been observed since May 2023, with current groundwater levels comparable to the water levels measured in 2002. EC values fluctuate between monitoring events with no observable trend. Ongoing monitoring will enhance data analysis for this borehole. There are currently no concerns.
G32933	18.088809	-32.99048	14	Lower Aquifer	0.3	68.1	The water levels are stable at ~3 mbgl. EC varies slightly between monitoring events, with a slight increase noted over the past month. GEOSS has sourced historical data for this site from 1981.

Borehole	Coordinates		Elevation (mamsl)	Aquifer	Collar height (m)	Borehole depth (m)	Field Comment
	East	South					
G33497	18.26099	-33.03743	104	Lower/ Deep Aquifer	1.39	60	Groundwater levels remain relatively stable at ~10 mbgl, with no signs of dewatering seen over the long term. Marginal fluctuations observed in water levels are a result of wellfield abstraction. EC measurements fluctuate within a stable range.
G33497A	18.26098	-33.03745	104	Upper/ Shallow Aquifer	1.25	32	Similar to G33497, groundwater levels are stable at this borehole at approximately 9 mbgl. The EC at this site fluctuates with a three-month average of 48.4 mS/m. No signs of dewatering are observed over the long term.
G33498	18.25110	33.07605	110	Lower/ Deep Aquifer	1.38	75	Groundwater levels remain relatively stable at this borehole. The groundwater EC at this site indicates natural fluctuations (48 mS/m to 75 mS/m). No concerns have been noted.
G33498A	18.25110	-33.07607	110	Lower/ Deep Aquifer	1.38	43	Manual hand readings over the long term do not display much of a response to the abstraction taking place at the Hopefield Wellfield. Groundwater levels have remained stable over the past year at approximately 22 mbgl. An overall improvement trend in EC has been observed since 2023.
G33501	18.30582	-33.07462	88	Upper/ Shallow Aquifer	1.38	45	Fluctuations observed in water levels are a result of abstraction taking place at the nearby Hopefield Wellfield. A rising trend in groundwater levels has been observed since February 2020, with current water levels comparable to those of 1987. EC measurements continue to fluctuate between site visits, which can be attributed to seasonal variation.
BG00168	18.32655	-33.05616	75	Bedrock Aquifer	0.49	31.5	A seasonal trend in water levels is becoming evident at this borehole. Electrical conductivity has continued to display an improvement over the past year. Ongoing monitoring will help identify long-term trends, which will be detailed in future reports. There are currently no concerns at this borehole.
GBK01	18.13189	-33.18704	2	Upper/ Shallow Aquifer	-	13	Groundwater levels and electrical conductivity values remain relatively stable over the long term. Fluctuations in water levels and EC are a result of seasonal variation and tidal influence. The EC values fluctuate between 85 mS/m and 160 mS/m. Monitoring data show no indication that wellfield abstraction or upgradient mining activities are adversely affecting this borehole.
GBK03	18.14594	-33.18171	34	Lower/ Deep Aquifer	0.2	60	Groundwater levels are stable at this borehole over the long term. The EC at this borehole fluctuates within a stable range, with no observable trend. Fluctuations are due to natural variation. There is no evidence that wellfield abstraction or upgradient mining activities are having any negative impact on this borehole.

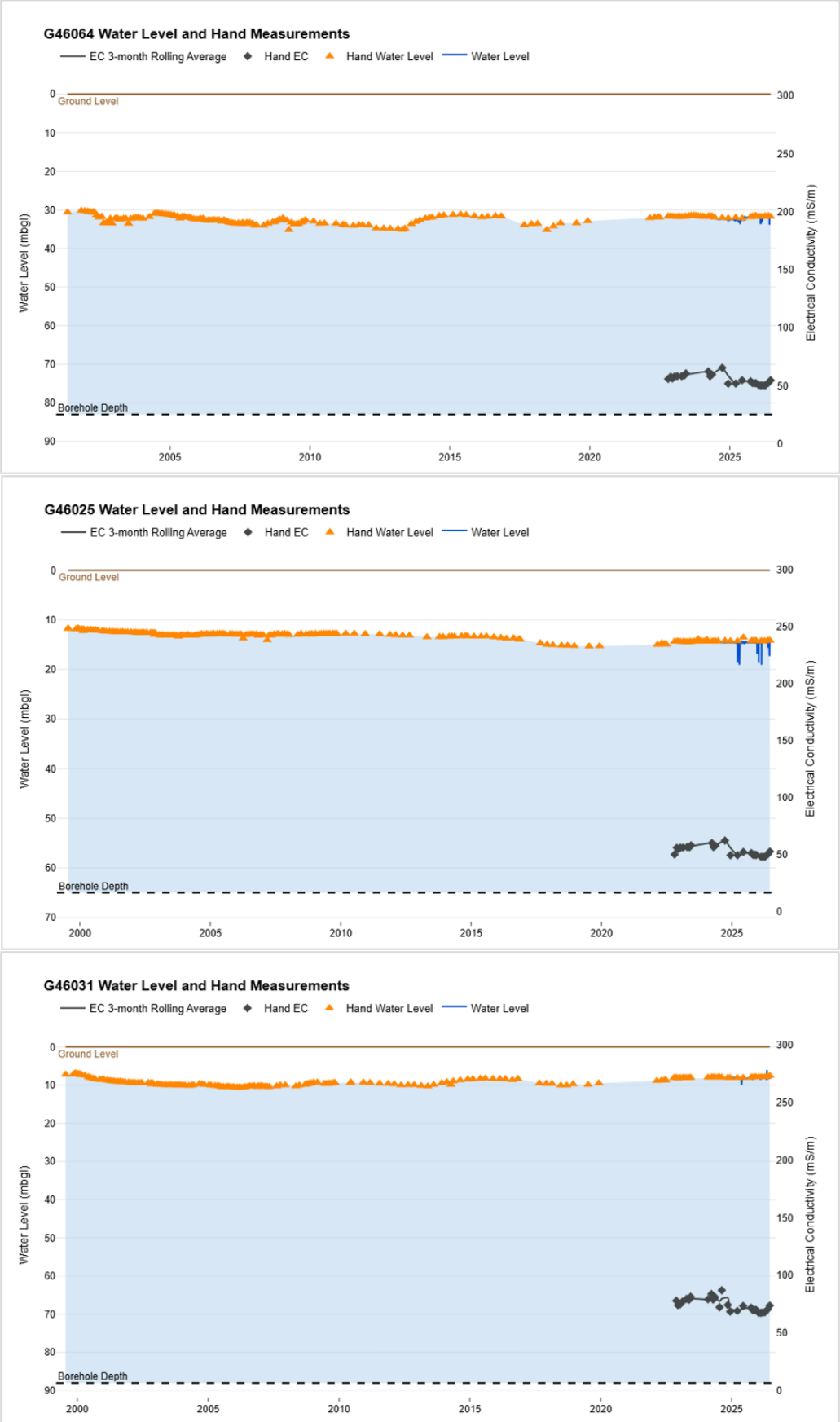

Figure 13.4. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

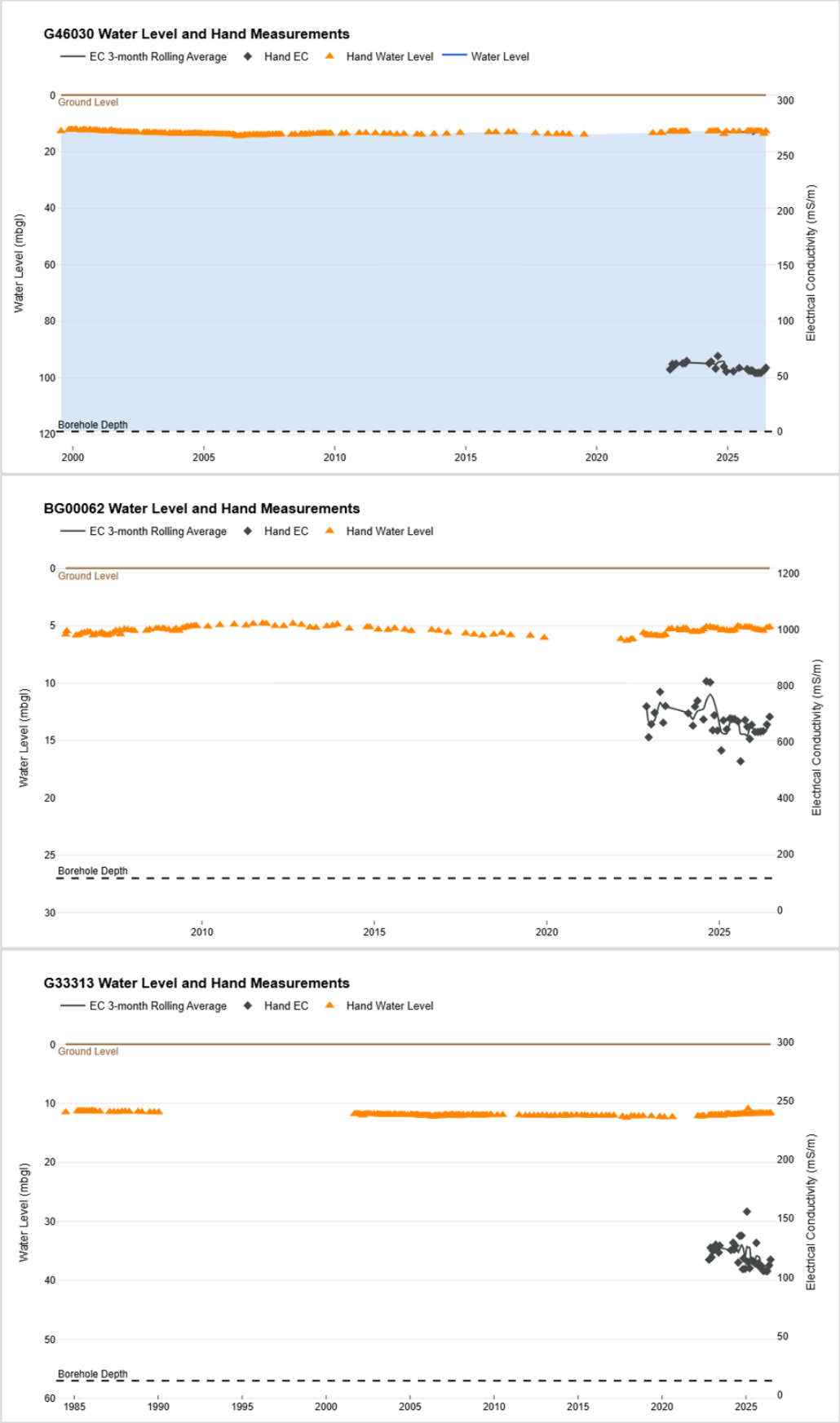

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

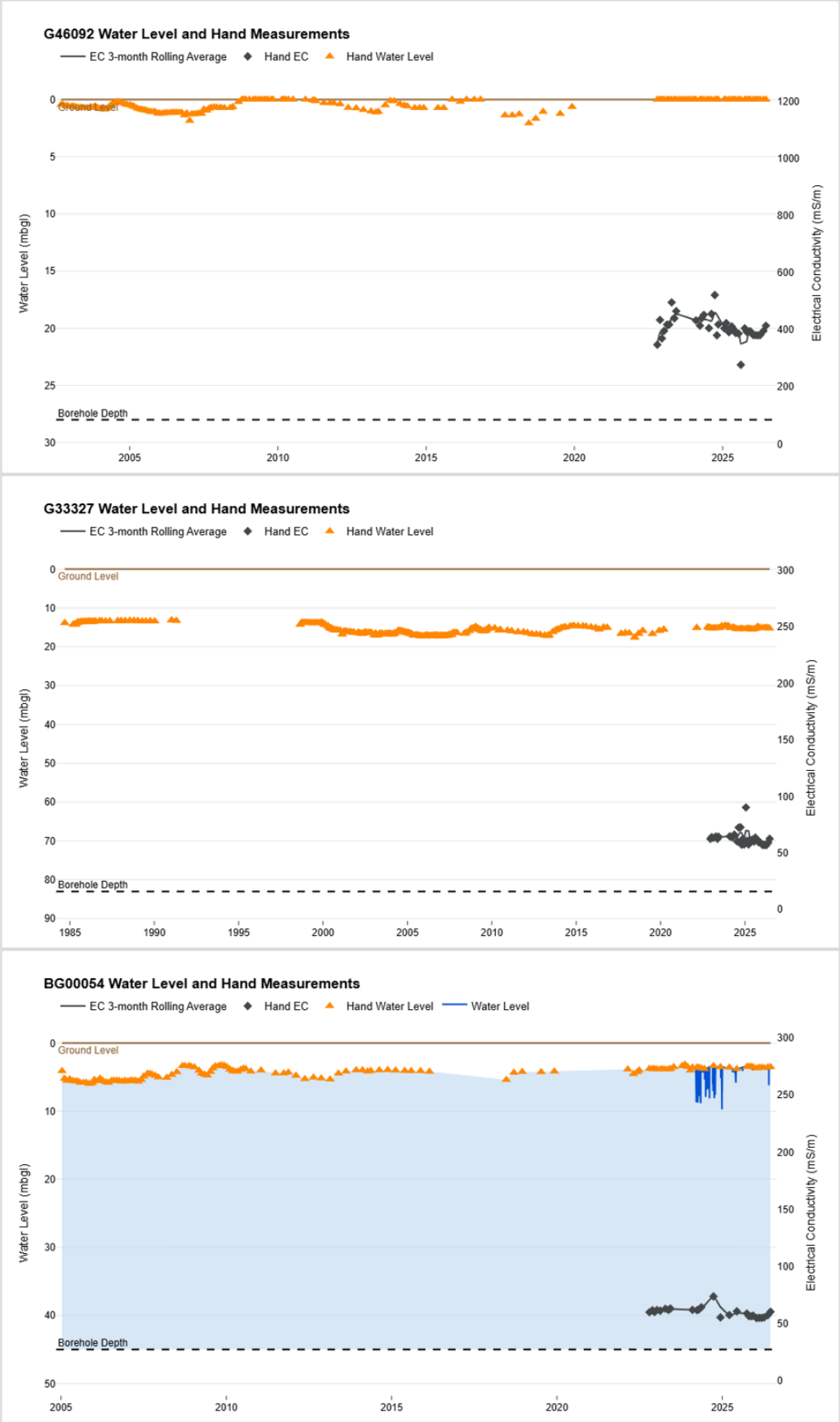

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

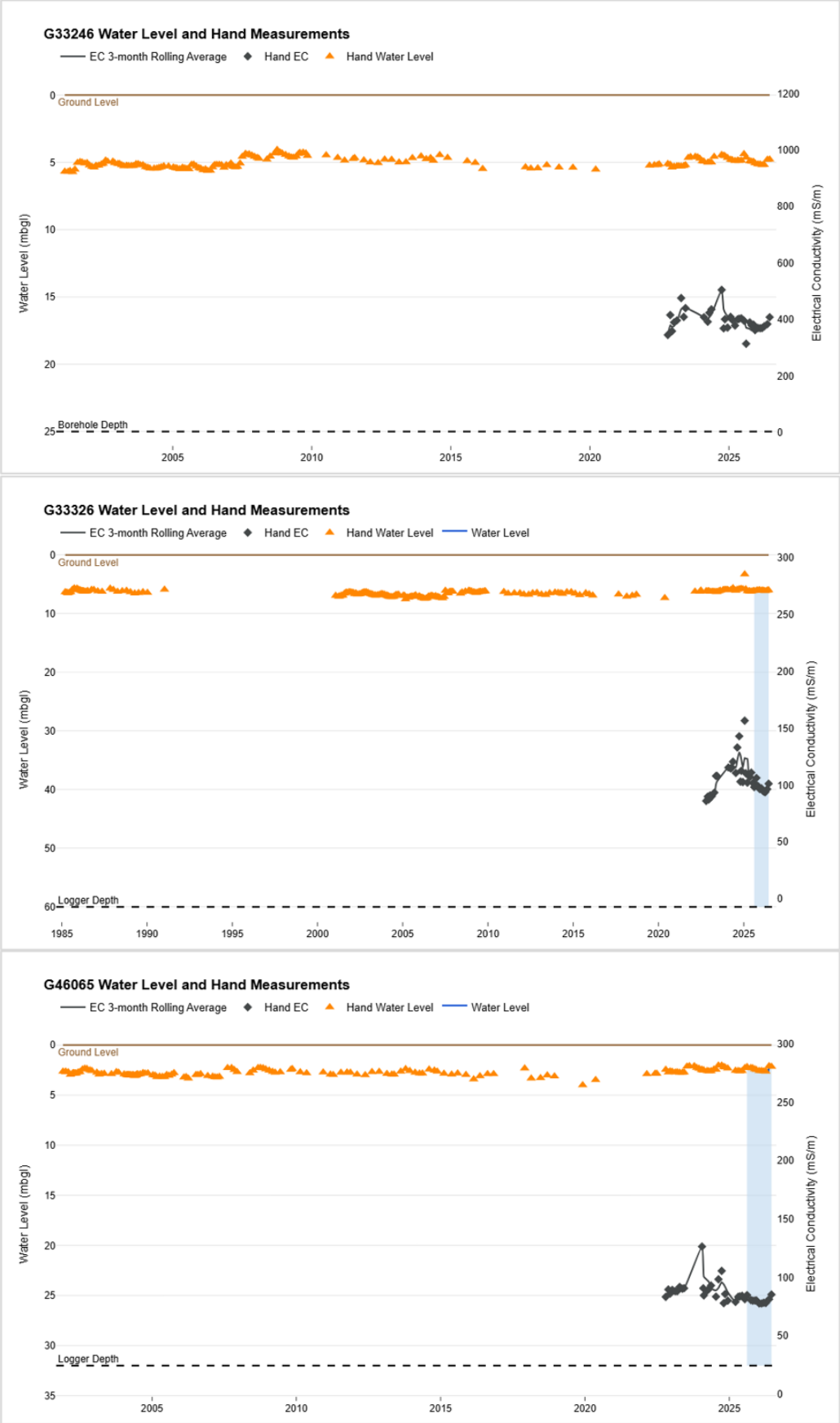

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

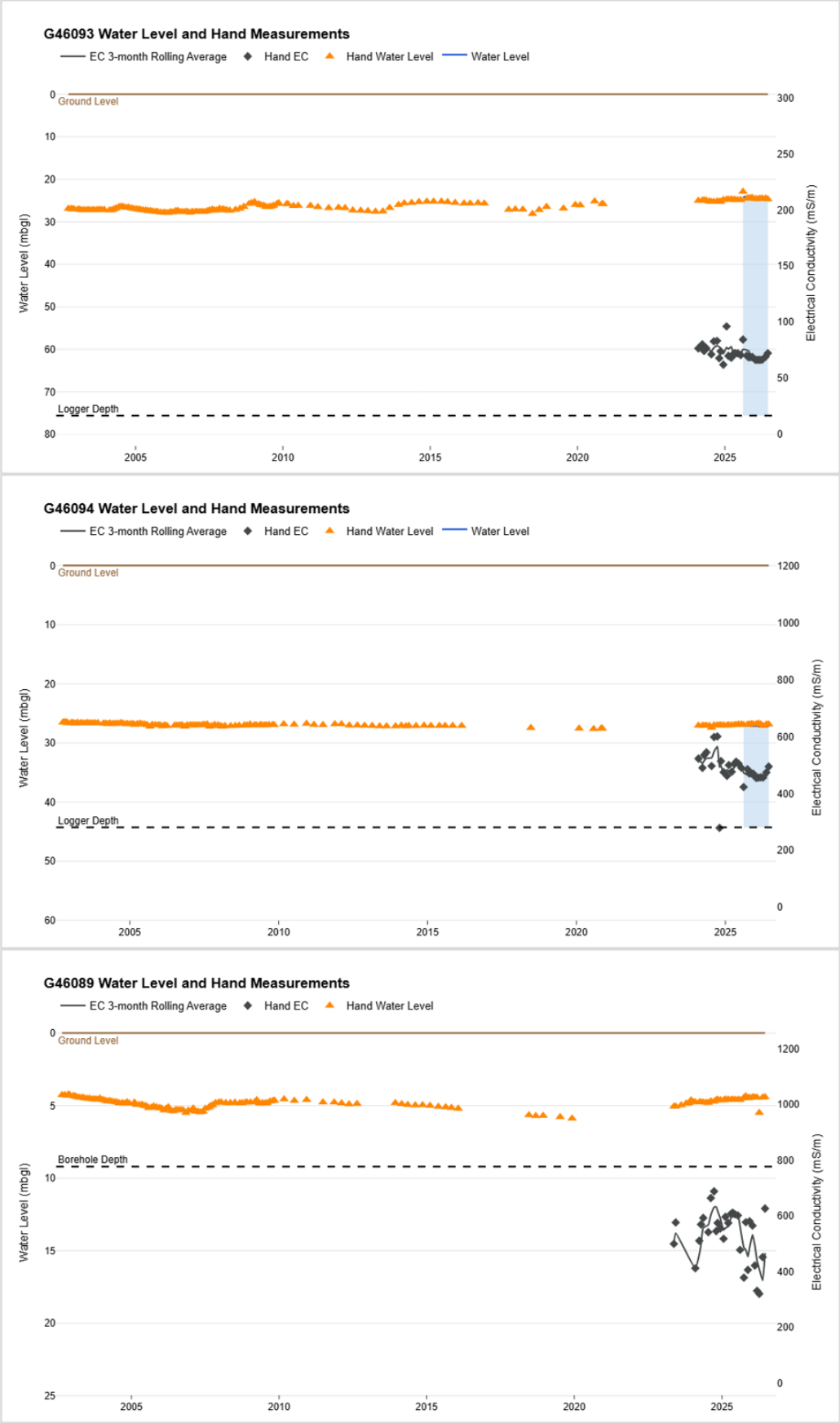

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

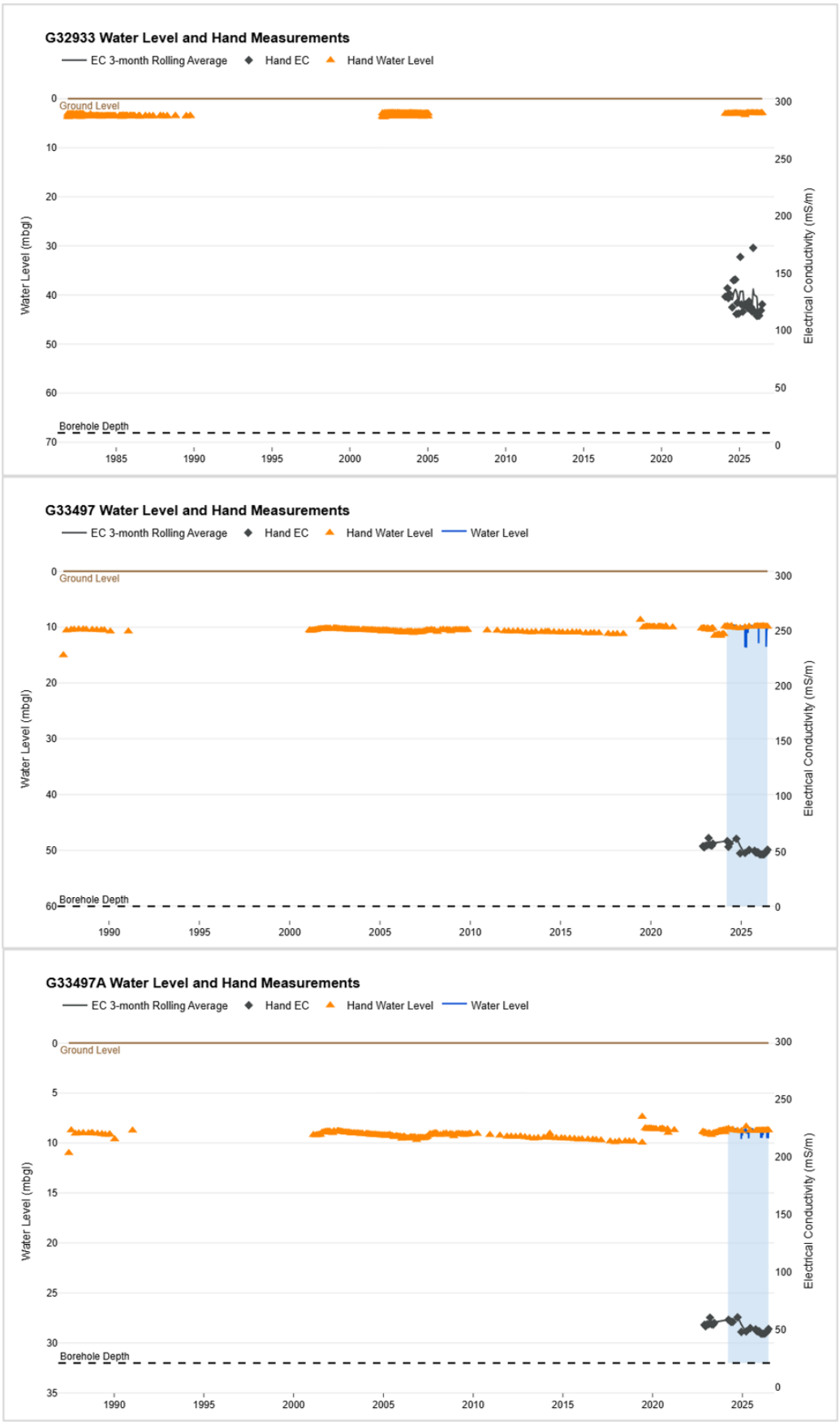

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

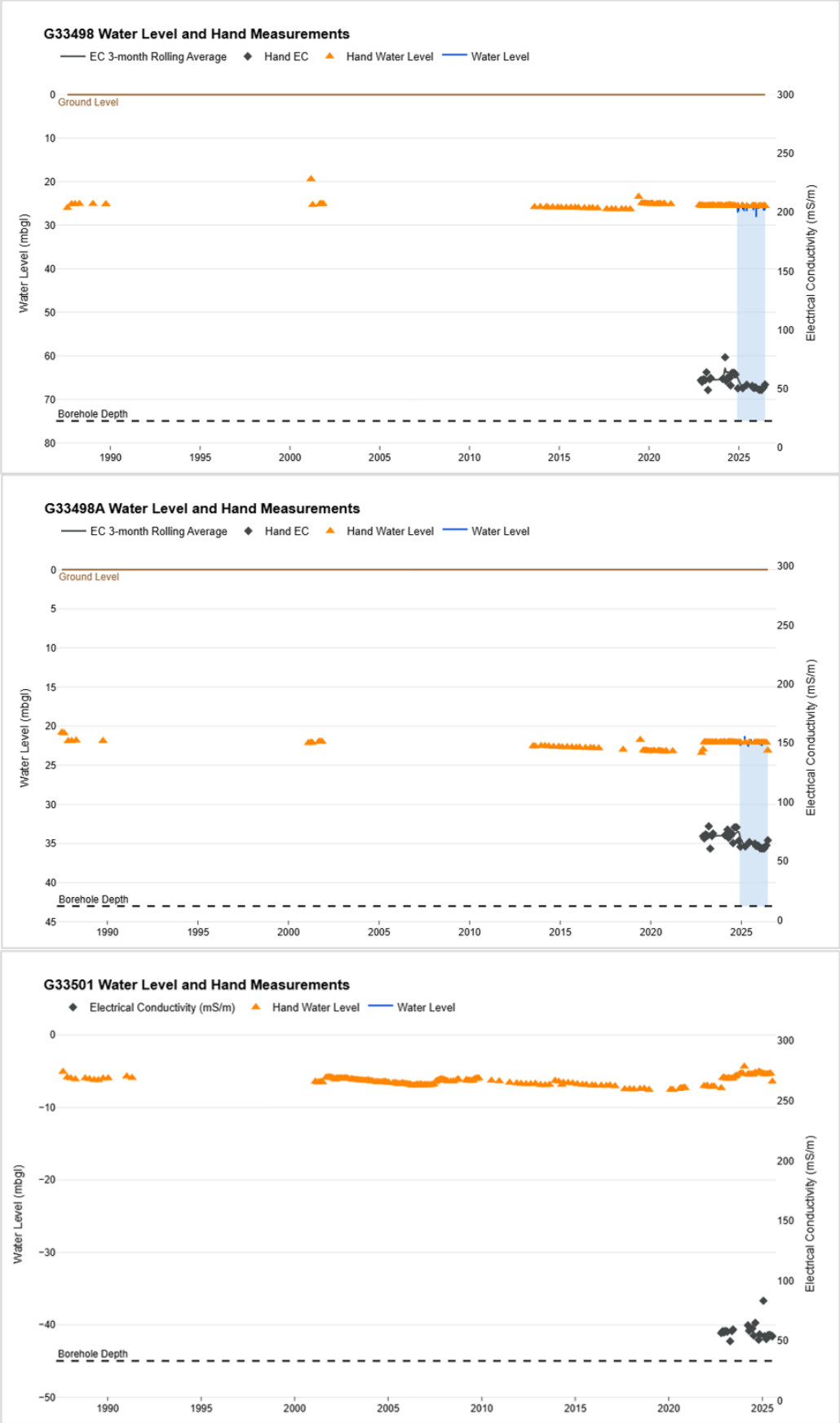

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

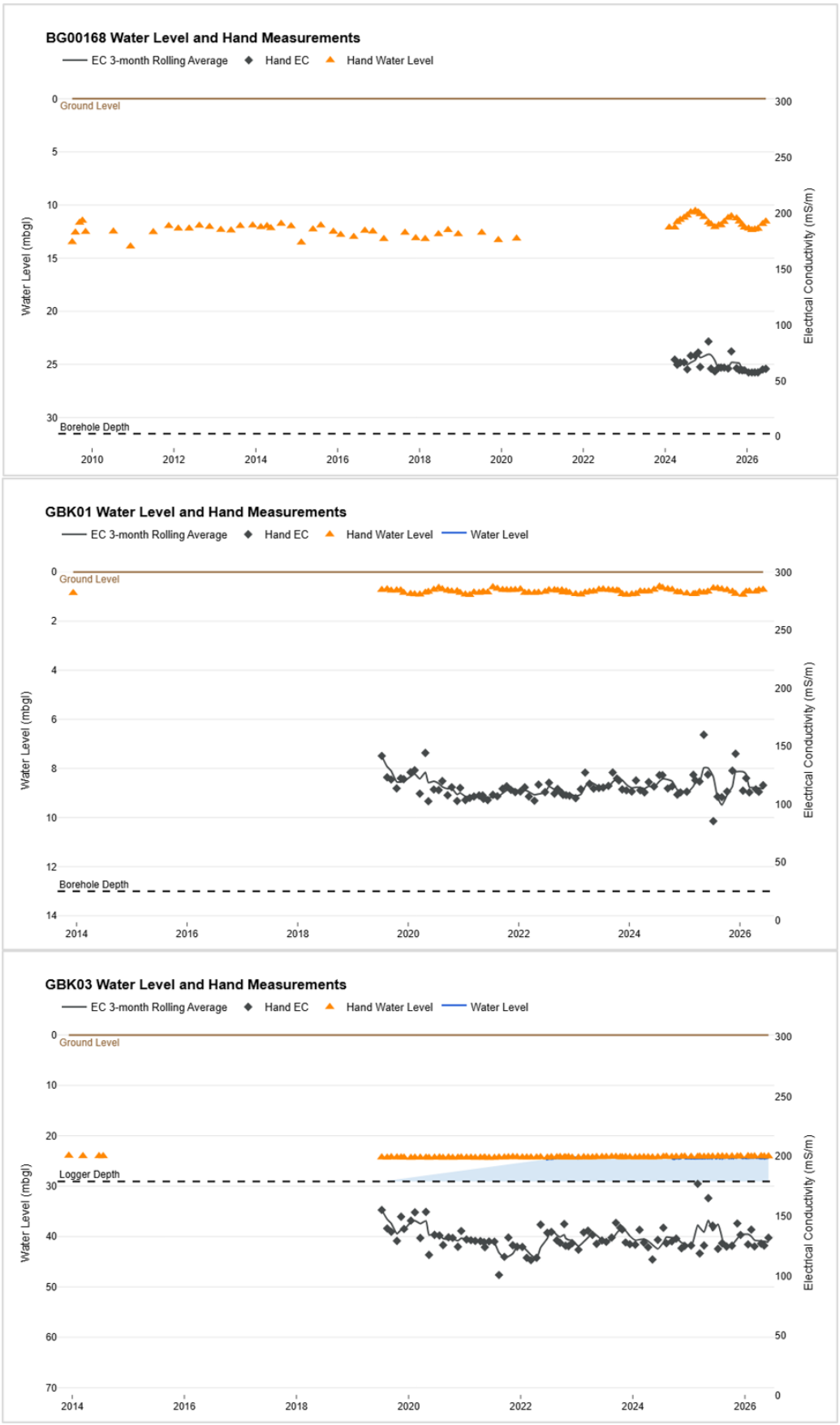

Figure 13.4 cont. Long-Term Water Level and Electrical Conductivity of Saldanha Bay Local Municipality Monitoring Boreholes (source: GEOSS 2026).

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