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SALDANHA BAY SEA BASED AQUACULTURE DEVELOPMENT ZONE

ANNUAL BENTHIC CHEMICAL SURVEY REPORT

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SALDANHA BAY SEA BASED AQUACULTURE DEVELOPMENT ZONE

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June 2024

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EXECUTIVE SUMMARY

Saldanha Bay is the primary area for bivalve production in South Africa, with the majority of national oyster and mussel production to date originating here. Monitoring of benthic impacts below mariculture installations is international best practice and has been undertaken in Saldanha Bay to validate dispersion model predictions of minimal impact since the Environmental Authorisation was granted. The Branch Fisheries Management appointed an independent specialist to compile a Sampling Plan for the ADZ which was reviewed by local and international stakeholders and experts (DAFF, 2018). This plan has since been updated and amended in 2022 following the outcomes of monitoring studies. Anchor Research and Monitoring (ARM), were appointed to undertake the annual benthic chemical monitoring surveys in adherence to this sampling plan, as supported by the branch Fisheries Management of the Department of Forestry, Fisheries and the Environment (2019-2020 & 2024-2027) and the World Wildlife Fund (WWF) South Africa through its Fish for Good initiative (2021-2023).

There are a wide range of benthic indicators in use by different countries, but they all have primary Environmental Quality Objectives of preventing hypoxic or anoxic sediment conditions by maintaining a functional benthos beneath the culture structures. Organic deposition and the subsequent decomposition by sediment bacteria increases oxygen demand which can lead to anaerobic conditions in the porewaters of the seabed beneath both finfish and shellfish farms. In severe cases this can lead to oxygen depletion in the water above the sediments, which may have direct impacts on farm operations. 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 and monitoring adverse impacts below finfish aquaculture. The inversely related chemical indicators Eh and S^{2-} have been used to classify sediments associated with fish farming into five organic enrichment groups: two oxic, two hypoxic and one anoxic. The Aquaculture Stewardship Council (ASC, 2017) specifies a S^{2-} thresholds of < 1500 µM (or Eh > -50 mV) as the threshold target beyond the Acceptable Zone of Effect (AZE). The benthic AZE is defined as 30 m from a fish cage array unless a site-specific zone of impact has been established. The Saldanha Bay ADZ Protocols for Environmental monitoring (commonly referred to as the Sampling Plan) proposed that this threshold is adopted for Saldanha Bay fish farm sites as the threshold outside the AZE. An additional S^{2-} threshold concentration of >3000 µM (or Eh < -100 mV) should be applied at the position of the finfish cages (DAFF, 2018). For shellfish aquaculture sites the Sampling plan recommended that S^{2-} threshold concentration of >3000 µM (or Eh < -100 mV) be adopted for annual monitoring of site condition in the shellfish aquaculture zones (ASC, 2012). Failure to meet S^{2-} thresholds of 1500 µM (Eh of -50 mV) at the AZE limit for finfish farms or 3000 µM (Eh of -100 mV) 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 stations.

There has, however, been some recent research on the measurement of total dissolved sulphides in organically enriched marine sediments below aquaculture infrastructure. Two studies demonstrated that the commonly used ion-selective electrode method for determination of free sulphides in a sediment slurry can lead to significant positive bias (Brown *et al.* 2011, Cranford *et al.,* 2020). Brown *et al.* (2011) reported orders of magnitude higher sulphide concentration detected in the buffered sediment–porewater slurry using the ion-selective electrode method than in porewater samples isolated and analysed separately using the methylene blue method (as used in this study). Cranford *et al.* (2020) compared three methods of measuring sulphide in marine sediments (methylene blue colorimetric, direct ultraviolet spectrophotometry and ion selective electrode) and found good agreement between the former two methods and the same positive bias with the latter method. These authors empirically compared the relationships between total free sulphide in marine sediment (measured using direct ultraviolet spectrophotometry) with several macrofauna indicators and developed a set of revised Ecological Quality Status (EQS) categories. For this study, sulphide concentration was determined by the Council for Scientific and Industrial Research (CSIR) using the methylene blue colorimetric method and we have applied the revised EQS categories to the interpretation of sulphide results, rather than the equivalent Hargrave *et al.* (2008b) geochemical categories.

Sediment was successfully collected from 38 sites within the ADZ during March 2024. This included 31 diver collected sites, 19 Sites in Big Bay (including two new sites), and six within the Small Bay centre precinct and six newly sampled sites within the Small Bay Mussel Raft Precinct. In addition, seven sites were sampled in Outer Bay North using a Van Veen Grab to limit risks to divers due to increase sampling depth. Triplicate redox and sulphide samples were analysed for each site.

Divers noted a strong odour while underwater collecting samples under the Small Bay Mussel Raft sites. On visual inspection four samples showed varying degrees of dark/black colour as well as having a strong sulphur odour emanating from the sediment when the sample jars were re-opened for Redox analysis. While most samples were fine mud to sandy samples, some include course sand and shell matter.

All finfish sites in Big Bay, except one, were not significantly different from the prescribed threshold (no finfish aquaculture is underway in the Bay). While the FF 60 m site was significantly lower (1-sample t-test: $p = 0.048$) than the specified -50 mV finfish threshold, no Big Bay farm or finfish sites differed statistically from the Big Bay control redox values, and are therefore not of concern. The average Redox potential for the Big Bay Finish sites was lower than that of the Big Bay Shellfish farm sites, which was in turn lower than the Big Bay control sites, suggesting a Bay wide reduction in Redox values which is likely due to natural geochemistry.

The majority of the Sulphide values for the Big Bay farm sites were significantly lower than the prescribed threshold and although sites B 4 and B 10 both differed significantly from the threshold value, they did not differ significantly from the reference sites. Similarly, Sulphide readings at finfish sites were not significantly higher than the prescribe threshold.

Redox potential in Outer North Bay varied, with positive and negative values. However, none of the farm sites were significantly lower than the prescribed threshold. Similarly, although three high sulphide values were recorded, two were farmed sites and one a control site. Only one of these average sulphide values (NB 3) significantly exceeded the prescribed threshold and yet it did not differ significantly from control sites. With the exception of NB 3, which also had a high Sulphide reading last year, no distinct pattern was visible amongst the years of available Sulphide data, and with no statistical differences between farmed and control sites in Outer Bay North there is presently no reason for concern.

Average redox values recorded within the centre Small Bay precinct all had negative readings and all except SB 2 exceeded the Hypoxic B threshold value of -100 mV. However, none of the average Redox concentrations were significantly lower than the threshold value or significantly different from the control sites. Similarly, all Sulphide values in the centre Small Bay precinct did not differ statistically from the threshold or the control sites. Therefore, there is no need for management action in this portion of the precinct.

Sites under the mussel rafts located on the southern boundary of Small Bay were sampled for the first time in 2024. It was below these rafts that divers mention smelling a strong, unpleasant odour while underwater collecting the samples. In addition, a number of these samples exhibited visual discolouration and further odour during the onshore redox analysis. SBM 1 and SMB 2 both showed high redox and sulphide values, however, only the redox values differed significantly from the prescribe threshold value, and neither the redox, nor the sulphide values from farmed sites were statistically different from the control sites.

The Big Bay site, B 4, previously flagged as being of concern due to high Sulphide and Redox values in 2021 and 2022, had significantly lower values in 2023 than in previous years and fell within the Moderate to Good categories. Suggesting that previous high results may have been the result of temporary and variable sediment and organic matter deposition. And no further management action is required. Despite this, it is still advised that sites B 9 and B 10 should still be sampled in the next annual benthic chemical survey to address the spatial gap in sampling sites downwind of mariculture infrastructure. Despite this given the strong odour (even underwater) and sediment discolouration of these samples it is suggested that these sites be monitoring in future and if they continue to show high levels of redox and sulphides that management actions may be needed.

Eight new sites were included in the sampling survey in 2024, two in Big Bay (B 9 and B 10) and three farm and three reference sites under and surrounding the mussel rafts on the southern boundary of Small Bay. Of these, B 10, SBM 1 and SBM 2 all exceeded the threshold values for both redox and sulphides.

The Big Bay site, B 4, previously flagged as being of concern due to high Sulphide and Redox values in 2021 and 2022, had significantly lower values in 2023 than in previous years, but was again high in 2024, falling in the 'Bad' categories, although several control sites also exceeded the threshold suggesting reduced conditions bay-wide.

It has been highlighted that 2024 Big Bay samples that exceeded the Anoxic redox potential threshold were all in water depths greater than 15 m. This suggests that the organic matter from the aquaculture development sites may pools in deeper areas with the Bay, causing these sites to have lower redox potentials. This hypothesis should be checked during future surveys, and should this pattern recurred, it may be necessary to add an additional control site in shallower water located on the eastern boundary of the Big Bay ADZ precinct.

Recommendations for future monitoring are provided below and should be incorporated into amendments/ updates to the Sampling Plan. The following provides a summary of key findings from the 2024 chemical survey:

The following provides a summary of key findings from the 2023 chemical survey:

1. Analytical laboratory measurements of sulphide concentrations in sediments were undertaken during the 2024 survey. Recent research indicates that the methylene blue method employed by the contracted laboratory (CSIR) results in sulphide measurements that are considerably lower (and more accurate) than those obtained using and ion-selective electrode protocol (upon which the Sampling Plan (2018) and Hargrave *et al.* (2008b) Geochemical categories are based). The recent 2021-2024 surveys used the former methodology and thus it is recommended that future ADZ monitoring continue to use either the ultraviolet spectrometry or the methylene blue methods of sulphide measurement and the revised EQS categories (Cranford *et al.* 2020) to assess sediment health below mariculture facilities.
2. Redox potential measurements are relatively inexpensive and easy to obtain and should continue to be collected alongside sulphide measurements to provide additional information on the state of the benthic environment and allow for comparisons with redox measurements taken to date.
3. It is recommended that, when possible, divers are used in preference to grab sampling for the collection of sediment samples as this carries a lower risk of oxidation. However, the collection of Outer Bay North samples using a Van Veen Grab, to reduce the inherent risk associated with greater sampling depths, has not resulted in any observable differences in sample results.
4. In instances where farming structures fall over hard substrata, redox and sulphide measurements are not considered suitable tools for monitoring the health of the benthic environment as sediment cannot be collected from hard substrata (this was the case with many of the FF stations in 2022 and the course sand/shell grit available at these sites often displays reduced results. It is still advised that alternative means for monitoring the health of the benthic environment in these areas. Assessments of visual or photographic reef quadrats will be undertaken in the 4th quarter of 2024 and a revised Bathymetry survey is scheduled form the 3rd quarter.
5. The substantially high redox and sulphide readings at some sites in the newly sampled Small Bay Mussel Raft precinct, along with in situ and laboratory observations of sediment odour and discolouration, suggest possible negative effects of the Aquaculture in the region. However, as these sites did not differ significantly from surrounding reference sites, it is merely suggested that they continue to be monitored. Additionally, the likely cause of these raised values (the lashing together of rafts due to mooring issues) has been rectified, and the practise is unlikely to continue going forward. Should similar conditions recur, management actions may need to be considered.

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1 BACKGROUND

The Branch Fisheries Management in the then Department of Agriculture, Forestry and Fisheries (now Department of Forestry, Fisheries and the Environment; DFFE), obtained Environmental Authorisation (EA) on 8 January 2018 to establish a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay. An ADZ is an area that has been earmarked specifically for aquaculture activities with the purpose of encouraging investor and consumer confidence, creating incentives for industry development, to provide marine aquaculture services, manage the risks associated with aquaculture, as well as to provide skills development and employment for coastal communities. The development of ADZs supports the Policy for the Development of a Sustainable Marine Aquaculture sector in South Africa (2007) objective aimed at creating an enabling environment that will promote growth and sustainability of the marine aquaculture sector in South Africa, as well as to enhance the industry's contribution to economic growth. The Branch Fisheries Management has created an enabling environment for the sustainable expansion within the ADZ operations in the existing aquaculture areas in Small Bay, Big Bay and Outer Bay North, and with the possibility to further extend operations into Outer Bay South/Entrance Channel. The authorized species for cultivation include both alien and indigenous species of finfish and shellfish, and seaweeds.

Saldanha Bay is the primary area for bivalve production in South Africa, with the majority of national oyster and mussel production to date originating here. As a result of improved opportunities for local mussel import substitution, the opening up of export markets for oysters, and improved access to water and land space through Operation Phakisa Oceans Economy, there is a renewed interest in expanding and fully utilizing the bay for further oyster and mussel production, as well as exploring potential finfish production in the outer, more exposed parts of the bay.

The then DAFF (now DFFE) appointed an Environmental Assessment Practitioner (EAP) to undertake an Environmental Impact Assessment for the establishment of an Aquaculture Development Zone in Saldanha Bay in 2016/2017. Appeals against the authorisation were lodged to the then Minister of Environmental Affairs and the authorisation was upheld as per the letter dated 7th June 2018. As required in terms of the EA, the Branch Fisheries Management appointed an Environmental Control Officer in 2018 and set up a Consultative Forum (CF – a public and industry forum), which includes approximately 127 members as of June 2024. The Aquaculture Management Committee (AMC – a government committee) meets every two months to ensure that the implementation of the ADZ occurs in line with the requirements specified in the EA and Environmental Management Programme (EMPr). The Branch Fisheries Management published a "Guideline for Bivalve Production Estimates for the Saldanha Bay Aquaculture Development Zone" in 2019, which has since been updated, and new guidelines added in the "Operational Guideline 2023: infrastructure in the Saldanha Bay ADZ". This document provides specific acceptable mussel and oyster infrastructure densities (mussel lines, oyster lines and/or mussel rafts) per precinct, along with infrastructure guidelines extracted from the ADZs EMPr that operators in the ADZ are required to uphold. Coupled with environmental monitoring, the adherence to the authorised tonnages should facilitate adaptive environmental management of the ADZ as a whole.

The Branch Fisheries Management appointed an independent specialist to compile a Sampling Plan for the ADZ which was reviewed by local and international stakeholders and experts (DAFF 2018). This plan has since been updated and amended in 2022 following the outcomes of monitoring studies.

Further work conducted for the ADZ by independent specialists include, dispersion modelling completed by PRDW, baseline macrofauna sampling done by Capricorn Fisheries Monitoring and macrofauna and physicochemical properties of the sediment analysed by Steffani Marine Environmental Consultant. In 2020, the Branch Fisheries Management appointed Anchor Research and Monitoring (ARM) to compile the ADZ baseline benthic survey report (Mostert *et al.* 2020a) and to conduct the annual redox survey and compile the resulting report (Mostert *et al.* 2020b). 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 ARM to undertake the 2021 benthic monitoring survey and conduct the annual benthic chemical surveys of the Saldanha Bay ADZ in 2021, 2022 and 2023 in an effort to support the development of the ADZ by fulfilling the requirements as per the Sampling Plan. To ensure the continued annual monitoring within the Bay, DFFE appointed Anchor Research and Monitoring (ARM) for the period of 2024-2027, to undertake the annual chemical survey sampling concurrently with the Annual State of the Bay sampling, a benthic monitoring survey, reef survey and Bathymetry survey. This report presents the findings of the 2024 benthic chemical survey undertaken during 11-15 March 2024.

2 INTRODUCTION

Monitoring of benthic impacts below mariculture installations is international best practice and is mandatory in all salmon growing countries (Black *et al.* 2008). Benthic monitoring is being undertaken in Saldanha Bay to validate dispersion model predictions of minimal impact (PRDW 2017, DAFF 2018). Although there is a wide range of benthic indicators in use by different countries, they all have primary Environmental Quality Objectives of preventing hypoxic or anoxic sediment conditions by maintaining a functional benthos beneath the culture structures (Black *et al.* 2008, PNS 2018). Maintaining functionality is crucial considering the importance of the benthos in promoting organic matter degradation by microbial communities.

Organic matter input from faeces, pseudo-faeces, uneaten feed and fall-off of culture organisms and fouling organisms is the primary source of impact on the seabed by aquaculture (Cranford *et al.* 2012, DAFF 2018). Shellfish feed on naturally occurring plankton populations which may result in an unnatural concentration of organic matter under farm infrastructure, however, this is typically of minor influence beyond the boundaries of the farm (NZMPI 2013). Generally, organic enrichment associated with bivalve aquaculture is less severe compared to finfish culture where artificial feed is used. Nevertheless, organic deposition and the subsequent decomposition by sediment bacteria increases oxygen demand which can lead to anaerobic conditions in the porewaters of the seabed beneath both finfish and shellfish farms (DAFF 2018). In severe cases this can lead to oxygen depletion in the water above the sediments, which may have direct impacts of farm operations as well as impacts on the benthic organisms. Ammonification and sulphate reduction to sulphides occur as typical responses to lowering of the oxygen reduction (Redox) potential (DAFF 2018). The production of sulphide by sulphate reduction is problematic, as sulphide is toxic (Black *et al.* 2008). However, it must be noted that highly organic enriched sediments can occur naturally where inputs from terrestrial or marine sources may be large, resulting in periodic oxygen depletion in sediments and overlying waters in these areas (DAFF 2018).

Sediment organic carbon, redox potential (Eh) and total sulphides (S^{2-}) have effectively been used in describing adverse impacts below finfish aquaculture (Hargrave 1994). Furthermore, the inversely related chemical indicators Eh and S^{2-} have been used to classify sediments associated with fish farming into four organic enrichment groups; normal, oxic, hypoxic and anoxic (Wildish *et al.* 2001). Oxic sediment typically has a high concentration of oxygen allowing aerobic respiration to occur, while in hypoxic conditions the amount of dissolved oxygen is limited but aerobic respiration continues although in a limited capacity (Diaz and Rosenberg 1995, Gray *et al.* 2002). Under anoxic conditions there is little to no oxygen available for aerobic respiration and anaerobic respiration takes over (Diaz and Breitberg 2009). Subsequently the classification was expanded into five groups with slight adjustments of the geochemical threshold levels, incorporating two oxic and two hypoxic categories as well as the anoxic category (Cranford *et al.* 2006, Hargrave *et al.* 2008a, Hargrave *et al.* 2008b). Each of the five defined categories has defined Eh and S^{2-} thresholds (Table 1). The inverse relationship between Eh and S^{2-} has proven to be comparable between both finfish and bivalve aquaculture sites (Cranford *et al.* 2006). Consequently, these chemical indicators provide an effective means of determining organic matter enrichment and oxic status of seabed deposits for both finfish and shellfish aquaculture operations.

There has, however, been some recent research on the measurement of total dissolved sulphides in organically enriched marine sediments below aquaculture infrastructure. Two studies demonstrated that the commonly used ion-selective electrode method for determination of free sulphides in a sediment slurry can lead to significant positive bias (Brown *et al.* 2011, Cranford *et al.* 2020). Brown *et al.* (2011) reported orders of magnitude higher sulphide concentration detected in the buffered sediment–porewater slurry using the ion-selective electrode method than in porewater samples isolated and analysed separately using the methylene blue method (as used in this study). Cranford *et al.* (2020) compared three methods of measuring sulphide in marine sediments (methylene blue colorimetric, direct ultraviolet spectrophotometry and ion selective electrode) and found good agreement between the former two methods and the same positive bias with the latter method. These authors empirically compared the relationships between total free sulphide in marine sediment (measured using direct ultraviolet spectrophotometry) with several macrofauna indicators and developed a set of revised Ecological Quality Status (EQS) categories (Figure 1).

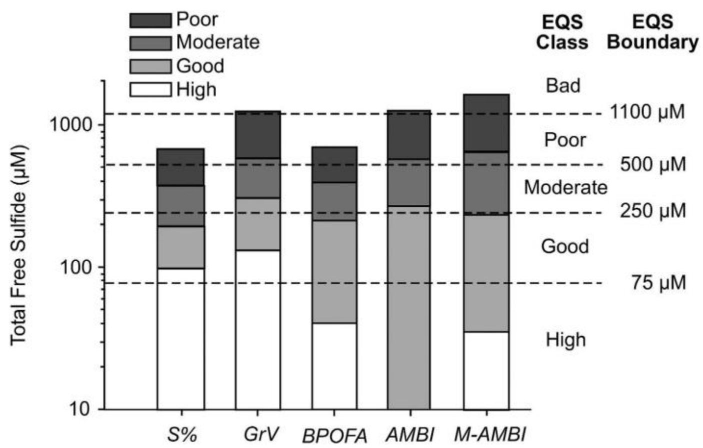


Figure 1. Total free sulphide concentrations and revised ecological quality status (EQS) boundaries for five benthic community indicators (Cranford *et al.* 2020).

The Sampling Plan identified the Aquaculture Stewardship Council's (ASC) thresholds as suitable for monitoring the impacts of finfish aquaculture in Saldanha Bay (ASC 2017, DAFF 2018). The ASC specifies a S^{2-} thresholds of < 1 500 µM (or Eh > -50 mV) as the target threshold beyond the Acceptable Zone of Effect (AZE). The benthic AZE is defined as 30 m from a fish cage array unless a site-specific zone of impact has been established. It has been proposed that this threshold is adopted for Saldanha Bay fish farm sites as the threshold outside the AZE. It has been suggested that an additional S^{2-} threshold concentration of >3 000 µM (or Eh < -100 mV) be applied at the position of the finfish cages (DAFF 2018). For shellfish aquaculture sites, it is recommended that S^{2-} threshold concentration of >3 000 µM (or Eh < -100 mV) be adopted for annual monitoring of site condition in the shellfish aquaculture zones (ASC 2012). Failure to meet S^{2-} thresholds of 1 500 µM (Eh of -50 mV) at the AZE limit for finfish farms or 3000 µM (Eh of -100 mV) 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 stations. For this study, sulphide concentration was determined by the Council for Scientific and Industrial Research (CSIR) using the methylene blue colorimetric method and we have applied the revised EQS categories to the interpretation of sulphide results, rather than the equivalent Hargrave *et al.* (2008b) geochemical categories (Table 1).

Table 1. Ranges of redox potential (Eh) and total sulphides (S^{2-}) in five sediment organic enrichment categories as indicated in the Sampling Plan (Cranford *et al.* 2006, Hargrave *et al.* 2008b, DAFF 2018) and recommended revised ecological quality standards (Cranford *et al*. 2020, DFFE 2022).

Geochemical	Oxic A	Oxic B	Hypoxic A	Hypoxic B	Anoxic
Ecological Quality Standard	**High**	**Good**	**Moderate**	**Poor**	**Bad**
Redox (Eh) mV	>100	100 to -50	-50 to -100	-100 to -150	<-150
Sulfides (S^{2-}) µM (Hargrave *et al*. 2008b)	<750	750 to 1500	1500 to 3000	3000 to 6000	>6000
Sulfides (S^{2-}) µM (Cranford *et al*. 2020)	<75	75-250	250-500	500-1100	>1100

3 METHODS

3.1 Sample collection

The annual redox survey of the Saldanha Bay ADZ was conducted during the annual Saldanha State of the Bay survey (11-15 March 2024). Sediment samples for the measurement of redox potential and sulphide (S^{2-}) were collected at 38 stations in Saldanha Bay (Figure 2, Table 2). This included, 10 impact sites, three control sites and six finfish sites (currently control sites as no finfish aquaculture is underway) in Big Bay. Two of these big Bay sites were sampled for the first time in 2024, as was recommended in the revised sampling plan. Three impact and three control sites were sampled in the centre of Small Bay, with Control Site 1 moved relative to previous years to ensure that the depth of this site is representative of other Small Bay sites. In addition, three new control and three new impact sites were sampled in the south of Small Bay, under the mussel rafts. Finally, four impact and three control sites were sampled in Outer Bay North (Figure 2, Table 2).

Figure 2. Map of Saldanha Bay showing the stations sampled during the 2024 annual benthic chemical survey of the Saldanha ADZ, control sites are indicated with blue arrows while impact sites are indicated with red arrows, and orange arrows for those added and newly sampled in 2024. Old sites replaced/repositioned are in grey.

Scientific divers collected triplicate sediment samples at each of the 31 stations. In the finfish precinct in Big Bay, three sediment samples were collected at 0 m, 30 m and 60 m along a transect from the edge of the proposed finfish cage location and three samples from within the precinct. Sediment samples were collected by the divers in new, 250ml polyethylene plastic jars which were sealed on the seafloor and then placed on ice aboard the survey vessel. Due to the increased depths of sampling sites in Outer Bay North (down to a maximum of 30 m), these samples were collected using a Van Veen Grab and bottled on the surface directly after being bought on board the boat.

Redox potential was measured using a Hach HQ 40 D portable meter equipped with an IntelliCAL® MTC101 ORP/redox probe. Measurements were conducted on the evening of the sample collection day immediately upon opening the sample jars. Photographs of the sediment samples were taken, and the sediment was observed for colour, visible out-gassing and smell. The sulphide samples were placed on ice until they were transferred to shore where they were frozen at -18°C until submission to the CSIR for sulphide (S^{2-}) analysis. The co-ordinates of the sites sampled are included in Table 2 below and shown on the map of Saldanha Bay in Figure 2.

Table 2. Co-ordinates of the chemical survey sites from Big Bay, Small Bay and Outer Bay North sampled in 2024, new sites are highlighted in red.

Area	Site	Latitude°	Longitude°	Comments
Big Bay	B 1	-33.028808	18.019161	
	B 2	-33.030550	18.022083	
	B 3	-33.039167	18.021183	
	B 4	-33.035367	18.010983	
	B 5	-33.044667	18.014917	
	B 6	-33.043950	18.009850	
	B 7	-33.031920	18.024640	These were moved in 2022 as previous sites were on the calcite reef, which hindered sampling.
	B 8	-33.028870	18.022320	
	B9	-33.034919	18.008291	New sites sampled as per 2022 revision of the sampling plan. First sampled in 2024
	B10	-33.032928	18.011506	
	BC 1	-33.029733	18.007400	
	BC 2	-33.048633	18.001550	
	BC 3	-33.065414	18.020089	
	FF 1	-33.039056	18.002878	
	FF 2	-33.040681	18.007119	
	FF 3	-33.042911	18.004736	
	FF Transect 0m	-33.042419	18.004349	
	FF Transect 30m	-33.042670	18.004450	
	FF Transect 60m	-33.042926	18.004562	
Outer Bay North	NB 1	-33.032617	17.943633	
	NB 2	-33.034417	17.948867	
	NB 3	-33.038433	17.945633	
	NB 4	-33.045200	17.942067	
	NB C 1	-33.037283	17.960267	
	NB C 2	-33.042167	17.953733	
	NB C 3	-33.03834	17.96395	
Small Bay	SB 1	-33.009100	17.964067	
	SB 2	-33.006717	17.967067	
	SB 3	-33.011133	17.969850	
	SB C1	-33.01139	17.981598	New site selected March 2022 for better depth representation
	SB C2	-33.006194	17.979093	
	SB C3	-33.010171	17.95587	
Small Bay Mussel Rafts	SBM_1	-33.028207	17.967333	New sites sampled as per 2022 revision of the sampling plan. First sampled in 2024
	SBM_2	-33.031754	17.970554	
	SBM_3	-33.035732	17.974049	
	SBM_C1	-33.038483	17.975748	
	SBM_C2	-33.025602	17.965655	
	SBM_C3	-33.023500	17.969833	

3.2 Laboratory analyses

Measurements of sulphide (S^{2-}) were undertaken by CSIR in Cape Town with reference to the Standard Methods for Examination of Water and Wastewater (4500-S2−SULFIDE, Methylene Blue Method). Pre-weighed wet sediment is acidified with Nitric acid (HNO_3) in an enclosed reaction vessel in the presence of continuous Nitrogen gas carrier. The liberated Hydrogen sulphide (H_2S) generated during the acidification is carried into receiving Zinc Acetate solution which converts H_2S into insoluble Zinc sulphide (ZnS) precipitate. The Sulphide is then quantified via iodometric titration and final result is based on the original mass of sample used (mg/kg and mmol/kg). There is a concern that the acid volatile sulphide methodology used would result in measurements of total sedimentary sulphide, including the chemically bound component (e.g. FeS) that is not bioavailable, rather than just free sulphide in pore water that is the ecotoxic component (Brown *et al.* 2011). However, this would result in overestimates of the free sulphur in samples and hence is a conservative approach (i.e. sulphide concentration results are likely to indicate poorer sediment quality than in reality). It is common practise that when a sample's value is below the laboratory detection limit, a value equal to half the detection limit is reported. In this case, numerous lab results indicated sulphide values were below the detection limit of 60 µM are therefore reported as 30 µM.

3.3 Statistical analyses

Survey results were tested for significant differences between chemical (redox and sulphides) sample and indicator thresholds (Table 1) and reference station average values according to statistical procedures given in the British Columbia Ministry of Environment protocols for marine environmental monitoring (BCME 2002). Univariate data were analysed using the software package, Dell STATISTICA v.13.

For finfish stations at 30m and 60m from the cages, samples were tested for chemical exceedance by a 1-sample t-test:

Redox: H_0: $\mu \geq -50$ mV; H_A: $\mu < -50$ mV (1-tailed)
Sulphide: H_0: $\mu \leq 250$ µM; H_A: $\mu > 250$ µM (1-tailed)

1. For stations at the fish cages (0 m) samples were tested for chemical exceedance by a 1-sample t-test:

   Redox: H_0: $\mu \geq -100$ mV; H_A: $\mu < -100$ mV (1-tailed)
   Sulphide: H_0: $\mu \leq 500$ µM; H_A: $\mu > 500$ µM (1-tailed)

   *Given that no finfish cages are presently located within the Bay – all finfish samples were tested against the -50 mV threshold.

   a) If there was evidence for exceedance at a particular station, a non-parametric 1-way Kruskal-Wallis ANOVA was performed to test if the values of farm (F) and reference stations (R) stations differed significantly:

      H_0: $\mu F \leq \mu R$; H_A: $\mu F > \mu R$ (1-tailed)

2. Samples collected at the shellfish farm site were tested for chemical exceedance by a 1-sample t-test:

   Redox: H_0: $\mu \geq -100$ mV; H_A: $\mu < -100$ mV (1-tailed)
   Sulphide: H_0: $\mu \leq 500$ µM; H_A: $\mu > 500$ µM (1-tailed)

   a) In the case of an exceedance, a nested non-parametric 1-way Kruskal-Wallis ANOVA was performed as above.

The redox and sulphide measurements are included in Appendix 1. Photographs of the sediment were taken and are included in Appendix 2.

4 RESULTS AND DISCUSSION

4.1 Nature of sediment

On visual inspection four samples showed varying degrees of dark/black colour as well as having a strong sulphur odour emanating from the sediment when the sample jars were re-opened for Redox analysis (Appendix 2). Namely, SMB_1 and SMB_2, B_4 and NB_3. Additionally, some samples were course and shelly – see BC_2, NB_C3 AND FF 30m (Appendix 2).

4.2 Big Bay

The majority of the Redox potential values for the Big Bay Shellfish farm sites were below the Anoxic limit, with the exception of B 4, B 7, B 9 and B 10, by contrast, two of the three Big Bay Control sites were above the Anoxic Limit of -150 mV (Figure 3). The only farmed/impact site that was significantly greater (more negative) than the accepted threshold value of -100 mV was B 4. However, non-parametric ANOVA shows that this value is not significantly different from those of the Big Bay control sites. Three sites within the finfish precinct were sampled (FF 1-3) as well as the three samples along a transect 0 m, 30 m and the 60 m. With the absence of any current finfish activity, these sites can all technically be recorded as control/baseline sites. Three of the six sites had positive redox values (FF 3, FF 0 m and FF 30 m), FF 1 was within the Hypoxic A threshold, with only FF 2 and FF 60 m recording average values greater than the Anoxic threshold (-150 mV). However, this was only significant at the FF 60 m site and none of the values differed statistically from the Big Bay control sites and are therefore, not of concern. (Figure 3). The average redox potential for all Finfish sites was -78.75 mV (moderate/Hypoxic A), the overall Big Bay farmed sites average was -150.5 mV (Poor/Bad), while the Big Bay Control sites average was over the threshold at -180.2 mV (Bad/anoxic) (see Table 1). This suggests that the low Redox potential is likely to be the result of the natural sediment geochemistry as the controls have the lowest average potential.

The majority of the Sulphide values for the Big Bay farm sites were significantly lower than the prescribed threshold of 500 µM (all were less than 130 µM), two sites (B 6 & B 8) did not differ significantly from the threshold and are all therefore of no concern (Figure 4). By contrast, sites B 4 and B 10 both differed significantly from the threshold. However, due to high variance within these sites, neither differed statistically from the Big Bay control sites. Sulphide readings at finfish sites were not significantly higher than the prescribe threshold (250 µM) and similarly did not differ significantly from the Big Bay Control sites (Figure 4).

In 2022, the redox measurement for site B 4 significantly exceeded the threshold specified for bivalve aquaculture (-100 mV) (1-sample t-test: $t = -29.40$, $p < 0.05$). A 1-way ANOVA was used to compare redox values at B 4 to the three reference stations in Big Bay as prescribed in the sample plan (DAFF 2018). ANOVA results indicated significant differences in redox values among these sites ($F_{3,8} = 51.16$, $p < 0.05$). Although no aquaculture infrastructure is located above B 4, it does lie to the South West of several longline installations and the prevailing winds during summer are southerly. In 2022 site B4 was therefore flagged as an area of potential concern, with suggestions that an additional 2-3 sites should be positioned in close proximity to this site of the Big Bay precinct to ascertain if the poor sediment quality at site B 4 is due to the bathymetry, or if there are wider spatial scale benthic impacts occurring downwind of the bivalve infrastructure. Although the revised sampling plan included the collection of samples at two (2) additional sites (B 9 and B 10), the 2023 sampling followed the 2018 sampling plan and did not include these two new sites. However, in 2023 both the Redox potential and Sulphide values for B 4 were significantly lower than in previous years and fell within the 'Moderate' to 'Good' categories. In 2024, the two new sites were sampled (B 9 and B 10) and redox and Sulphide values at B 4 and B 10 were greater than the prescribed threshold values. However, in no instances were these values significantly different from the Control sites.

It should be noted that all sites in 2024 that exceeded the Anoxic redox potential threshold (-150 mV) (B 4, B 7, B 9, B 10, BC 1, BC 2, FF 2 and FF 60 m) were in water depths greater than 15 m (15-17 m maximum) and were the deepest sites samples in Big Bay. Therefore, it is possible that the organic matter from the aquaculture development sites pools in deeper areas with the Bay, causing these sites to have lower redox potentials. This hypothesis should be checked during future surveys and should this pattern recurred it may be necessary to add an additional control site in shallower water located on the eastern boundary of the Big Bay ADZ precinct.

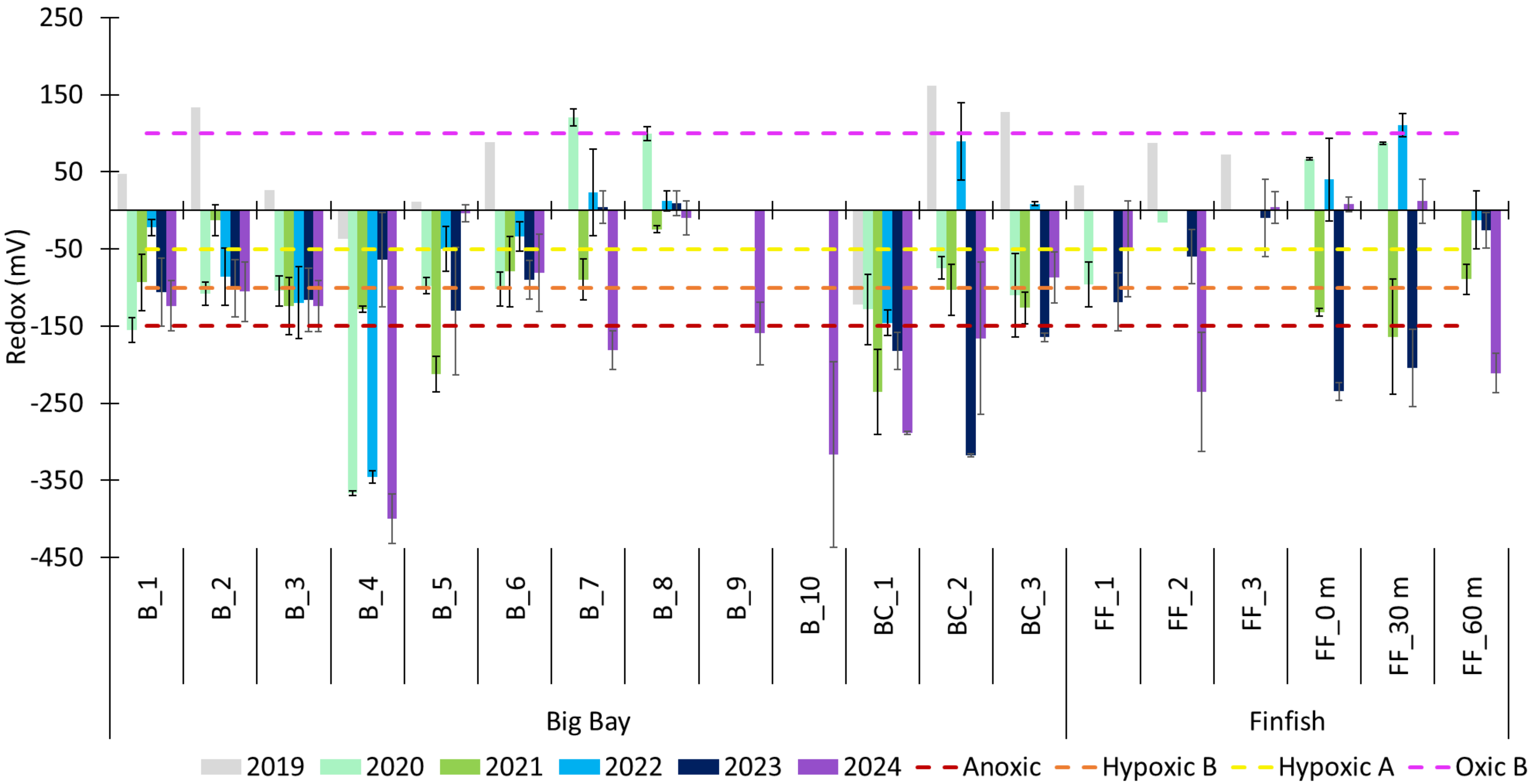


Figure 3. **Redox (mV) measurements recorded in <u>Big Bay</u> during the annual 2024 ADZ monitoring survey (bars ± standard error). Included are historical redox data sampled during the 2019-2023 surveys.**

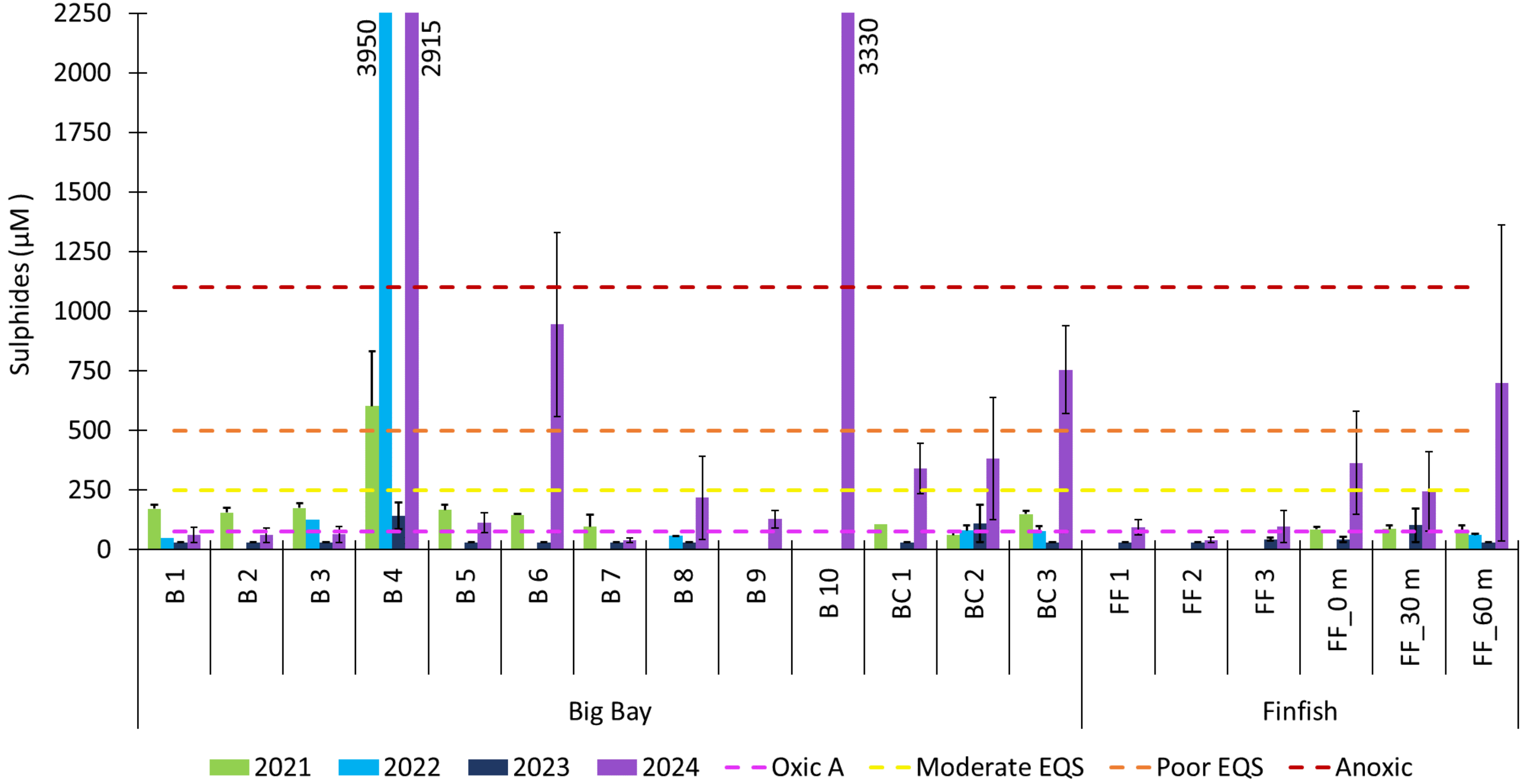


Figure 4. **Sulphide (µM) measurements recorded in <u>Big Bay</u> during the annual 2024 ADZ monitoring survey (bars ± standard error). Included are historical data sampled during the 2021-2023 surveys. Numbers next to bars represented continuation irrespective of designated vertical axis scale.**

4.3 Outer North Bay

Redox potential in Outer North Bay varied, with positive average values recorded at three sites (NB 4, NBC 1 and NBC 3), and negative values recorded at the remaining three farm sites and one reference sites (Figure 5). None of the farm sites were significantly greater (more negative) than the prescribed threshold of -100 mV and the overall average redox potential at the Farm sites (-87.64 mV) was more negative (moderate Ecological Quality Standard) than the average value in the reference/control sites (28.61 mV) which were in a Good Ecological Quality Standard (Figure 5, Table 1).

Conversely, the Sulphide value recorded in 2024 at NB 3 was significantly greater than the prescribe threshold (2162 ± 367 SE vs 500 µM), however, with a high control value at NBC 2 (1645 ± 186 µM) no farm site was significantly different from the Outer Bay North Control Sites (Figure 6). It is still worth noting that the average sulphide concentration for farmed sites was higher than that of the control sites (799 vs 587 µM) both of which fall within the Poor/Hypoxic B Ecological Quality Standard, making the average sulphide value in the entire North Bay in 2024 (708 µM) higher than averages seen in the bay in 2023 and 2022 (453 and 464 µM respectively). With the exception of NB 3, which also had a high Sulphide reading last year, no distinct pattern is visible amongst the years of available Sulphide data, and with no statistical differences between farmed and control sites there is presently no reason for concern.

The collection of these samples using a Van Veen Grab as opposed to divers, to reduce the inherent risk associated with greater sampling depths, has not resulted in any observable differences in sample results.

4.4 Small Bay (centre)

Average redox values recorded within the Small Bay lease area located within the centre of the Bay all had negative readings and all except SB 2 exceeded the Hypoxic B threshold value of -100 mV (Figure 5). However, none of the average Redox concentrations in the centre of the Bay were significantly lower than the threshold value (1-sample t-test) and they did not differ significantly from the redox values of the reference/control sites within Small Bay. Notably the redox values of the 2024 Small Bay sites were observed to be predominantly less negative than those recorded in 2023 (Figure 5). However, unlike in 2023 the overall average of the farm sites (274.27 mV) was more negative than that of the control sites (100.09 mV), although not significantly so.

All Sulphide values within the centre precinct of Small Bay did not differ significantly from the prescribe threshold (Figure 6). With an outlier in each of the farmed and control sites having high average sulphide readings but also high variance (SB 1 and SBC 3). Given that there were no significant differences between average Control and Impact sites for either Redox or Sulphide values in the centre of Small Bay, there is no need for management action in this portion of the precinct.

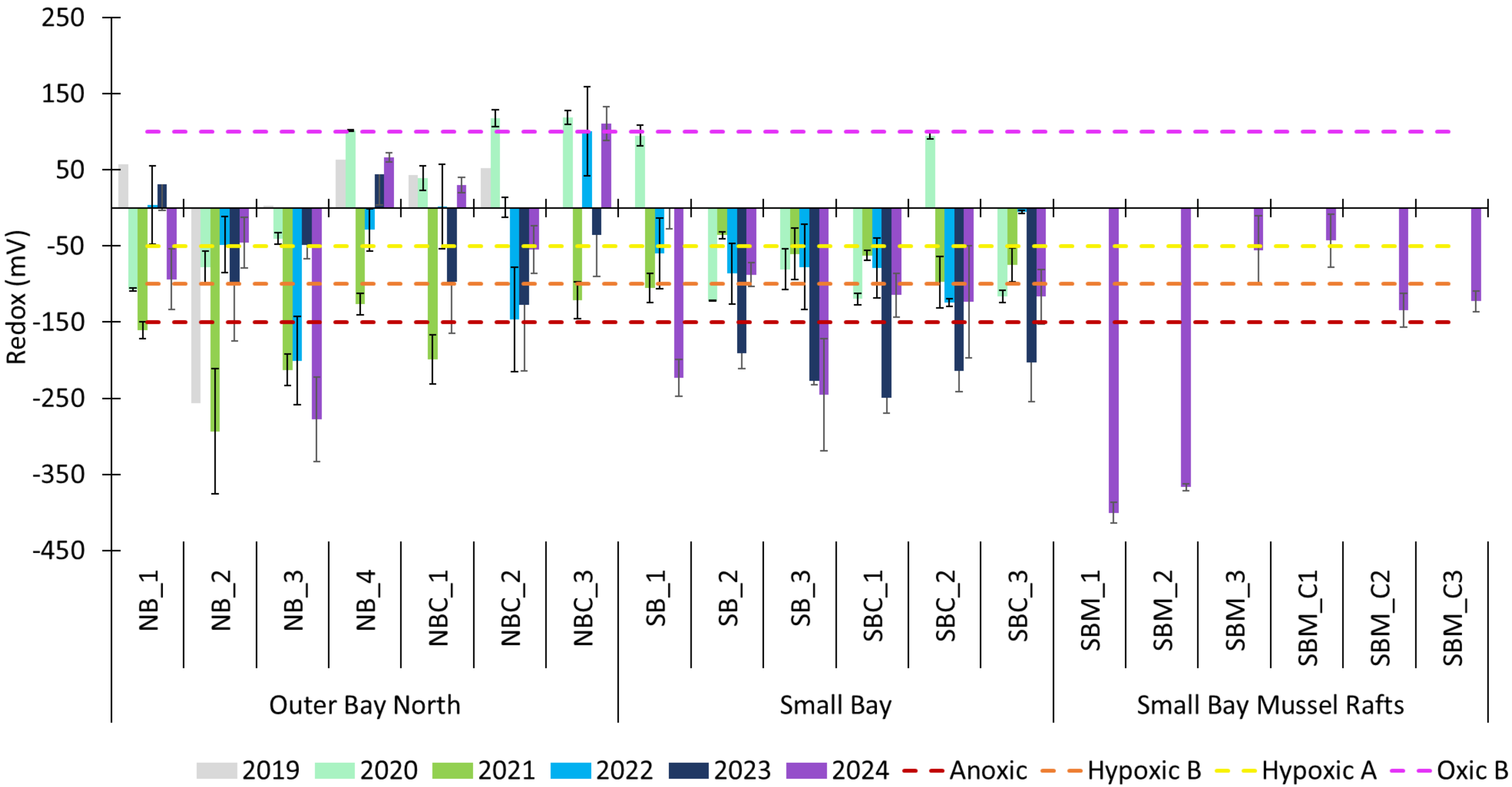


Figure 5. **Redox (mV) measurements recorded in <u>Outer Bay North and Small Bay</u> during the annual 2024 ADZ monitoring survey (bars ± standard error). Included are historical redox data sampled during the 2019-2023 surveys.**

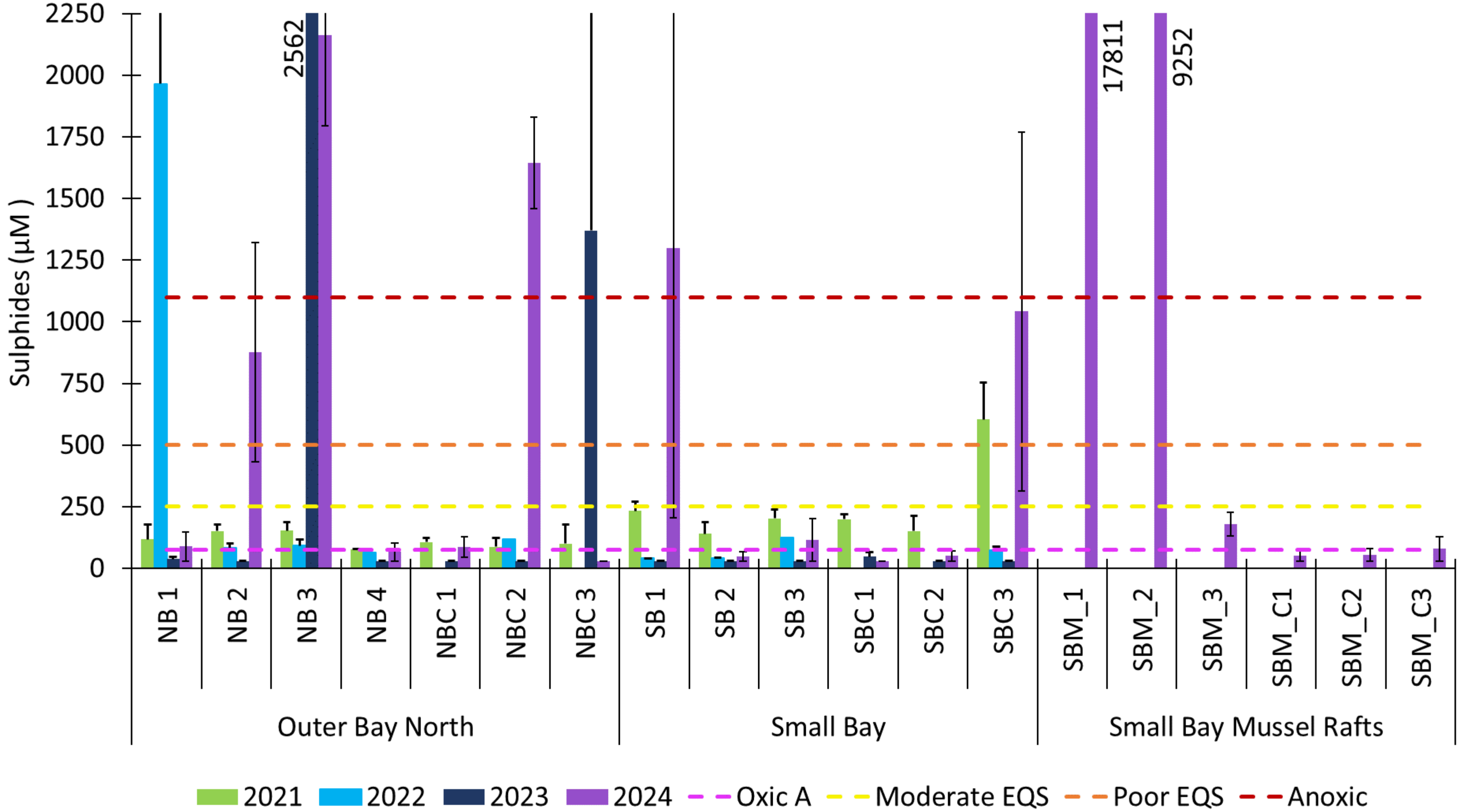


Figure 6. **Sulphide (µM) measurements recorded in <u>Outer Bay North and Small Bay</u> during the annual 2024 ADZ monitoring survey (bars ± standard error). Included are historical data sampled during the 2021-2023 surveys. Numbers next to bars represented continuation irrespective of designated vertical axis scale.**

4.5 Small Bay Mussel rafts

During the March 2024 survey, was the first year in which samples have been collected under the mussel rafts located on the southern boundary of Small Bay (Figure 2). The first notable factor for consideration is that the divers mentioned that they could smell a strong, unpleasant odour while underwater collecting the samples. Similarly, many of the samples shows a dark/black discolouration of the sediment (Appendix 2) suggesting anoxic conditions and also released a strong odour once reopened onshore to take the redox measurements.

SBM 1 and SMB 2 both showed high redox and sulphide values, however, only the redox values differed significantly from the prescribe threshold value, and neither the redox, nor the sulphide values from farmed sites were statistically different from those of the control sites (Figure 5and Figure 6). Despite this given the strong odour (even underwater) and sediment discolouration of these samples it is suggested that these sites be monitoring in future and if they continue to show high levels of redox and sulphides that management actions may be needed.

One should also note that Blue Ocean Mussels (the company operating in this precinct) has had rafts lashed together into larger ones for some time on that site, and so the raft size is basically double the normal raft size. The lashing is due to them having trouble detaching the moorings which were buried in the sand over time. They have since unlashed the rafts and according to the ADZ's Environmental Control Officer (ECO) all rafts have finally been secured to their own moorings except for two rafts which during the June ECO site visit were again lashed temporarily.

Many of these lashed rafts were located in the mid- and western portion of the precinct (where sites SBM 1 and SBM 2 are located). Therefore, the higher sulphide and redox readings in these areas may be due to the lashed rafts creating a greater impact zone than the other raft sites in the bay. Since this practise will no longer continue, as the rafts are now secured to their own moorings, we should see an improvement in the readings next year when sampling is repeated.

It was previously reported that sources of organic carbon and nitrogen in Small Bay; which include fish factory wastes, biogenic waste from mussel and oyster culture as well as sewage effluent from the wastewater treatment works, in conjunction with the sheltered nature of Small Bay have the potential to influence redox and sulphide readings and should be taken into account when assessing the future redox and sulphide measurements in this precinct (Mostert *et al.,* 2020a). In 2023, redox values in Small Bay were higher than all previous years. However, this was not the case for the centre sites in 2024, which were all fairly similar with no significant difference between farmed and reference sites. The exception was the Small Bay sites under the rafts, two of which showed high redox and sulphide values which far exceeded the prescribed limits, and yet, because of high variance they do not differ significantly from the controls and should only be monitored carefully going forward, with no direct action needed at present.

The entire Saldanha Bay, however, is a highly productive environment with considerable natural enrichment due to the advection of nutrient rich upwelled waters into the sun-warmed and relatively shallow bay. Seasonal (summer and autumn) natural hypoxia of deeper water is associated with upwelling processes and the decay of phytoplankton blooms, and this is reflected in the widespread negative redox values that were observed across all four lease areas within Saldanha Bay.

As in all previous years, the correlation between redox readings and measured sulphide concentrations was found to be poor (R^2 = 0.2326) (Figure 7). Other studies have also presented figures showing a particularly poor relationship for data in the negative redox potential range between sulphide concentration and redox protentional (e.g. Brown *et al.* 2011, Hamoutene 2014, Cranford *et al*. 2020). Therefore, although it is considerably more expensive, it is still recommended to measure both the Redox potential and the Sulphides within the sediments.

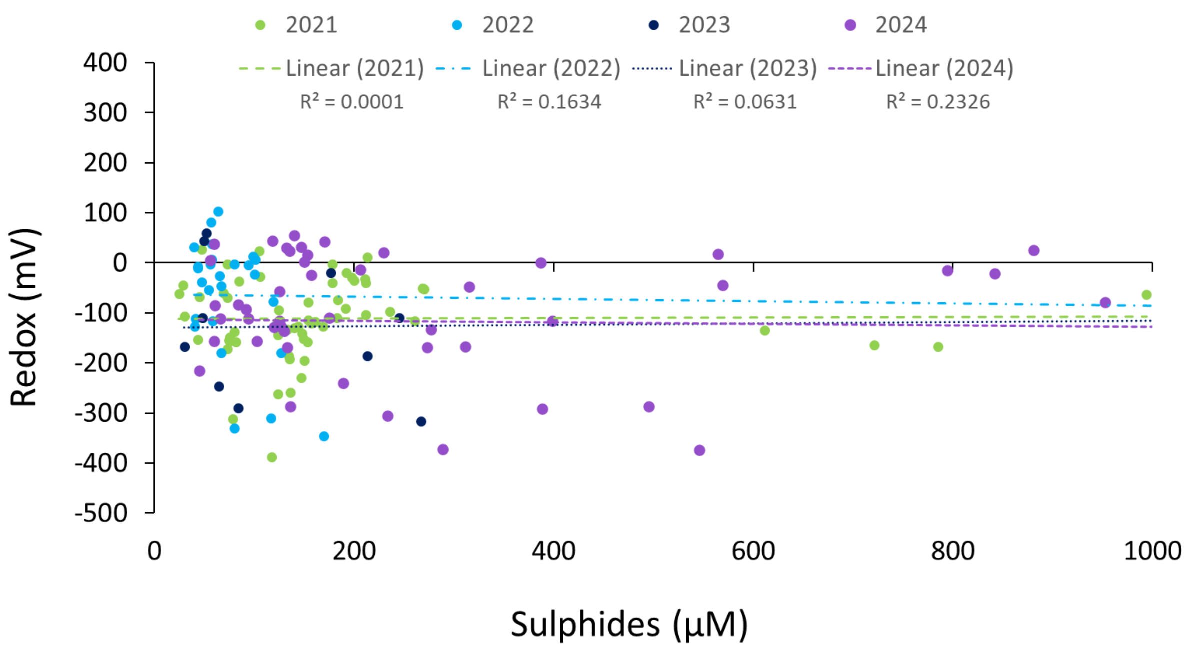


Figure 7. **Relationship between measured redox potential and sulphide concentration in sediment samples collected during the 2021-2023 survey of the Saldanha ADZ.**

5 FINDINGS SUMMARY & MANAGEMENT RECOMMENDATIONS

Overall, the redox values were reasonably consistent across the established ADZ lease areas with most site averages not significantly different from the prescribed threshold value (-100 mV) as stipulated by the Sampling Plan (DFFE, 2022). Any threshold exceedances that did occur were not significantly difference between farmed and reference sites, thus suggesting natural or bay wide impacts as opposed to specific ADZ related impacts. In 2024, more sulphide values exceeded the threshold than were seen in previous years. However, these occurred in both farmed and reference sites, which did not differ statistically.

Eight new sites were included in the sampling survey in 2024 (as recommended in the Sampling plan 2022), two in Big Bay (B 9 and B 10) and three farm and three reference sites under and surrounding the mussel rafts on the southern boundary of Small Bay. Of these, B 10, SBM 1 and SBM 2 all exceeded the threshold values for both redox and sulphides.

The Big Bay site, B 4, previously flagged as being of concern due to high Sulphide and Redox values in 2021 and 2022, had significantly lower values in 2023 than in previous years, but was again high in 2024, falling in the 'Bad' categories, although several control sites also exceeded the threshold suggesting reduced conditions bay-wide.

It has been highlighted that 2024 Big Bay samples that exceeded the Anoxic redox potential threshold were all in water depths greater than 15 m. This suggests that the organic matter from the aquaculture development sites may pools in deeper areas with the Bay, causing these sites to have lower redox potentials. This hypothesis should be checked during future surveys, and should this pattern recurred, it may be necessary to add an additional control site in shallower water located on the eastern boundary of the Big Bay ADZ precinct.

The following provides a summary of key findings from the 2023 chemical survey:

6. Analytical laboratory measurements of sulphide concentrations in sediments were undertaken during the 2024 survey. Recent research indicates that the methylene blue method employed by the contracted laboratory (CSIR) results in sulphide measurements that are considerably lower (and more accurate) than those obtained using and ion-selective electrode protocol (upon which the Sampling Plan (2018) and Hargrave *et al.* (2008b) Geochemical categories are based). The recent 2021-2024 surveys used the former methodology and thus it is recommended that future ADZ monitoring continue to use either the ultraviolet spectrometry or the methylene blue methods of sulphide measurement and the revised EQS categories (Cranford *et al.* 2020) to assess sediment health below mariculture facilities.
7. Redox potential measurements are relatively inexpensive and easy to obtain and should continue to be collected alongside sulphide measurements to provide additional information on the state of the benthic environment and allow for comparisons with redox measurements taken to date.
8. It is recommended that, when possible, divers are used in preference to grab sampling for the collection of sediment samples as this carries a lower risk of oxidation. However, the collection of Outer Bay North samples using a Van Veen Grab, to reduce the inherent risk

associated with greater sampling depths, has not resulted in any observable differences in sample results.

9. In instances where farming structures fall over hard substrata, redox and sulphide measurements are not considered suitable tools for monitoring the health of the benthic environment as sediment cannot be collected from hard substrata (this was the case with many of the FF stations in 2022 and the course sand/shell grit available at these sites often displays reduced results. It is still advised that alternative means for monitoring the health of the benthic environment in these areas. Assessments of visual or photographic reef quadrats will be undertaken in the 4th quarter of 2024 and a revised Bathymetry survey is scheduled form the 3rd quarter.
10. The substantially high redox and sulphide readings at some sites in the newly sampled Small Bay Mussel Raft precinct, along with in situ and laboratory observations of sediment odour and discolouration, suggest possible negative effects of the Aquaculture in the region. However, as these sites did not differ significantly from surrounding reference sites, it is merely suggested that they continue to be monitored. Should similar conditions recur, management actions may need to be considered.

6 APPENDIX 1

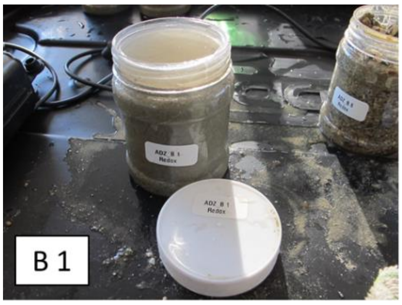
Table 3. Redox (mV) and Sulphides (µM) measured during the 2019 baseline survey and the 2020-2024 annual chemical survey. 2022 and 2024 measurements are continued on the following page.

Area	Site	2019	2020			2021						2022			2022
		Redox	Redox 1	Redox 2	Redox 3	Redox 1	Redox 2	Redox 3	Sulph 1	Sulph 2	Sulph 3	Redox 1	Redox 2	Redox 3	Sulph 1
Big Bay	B 1	47.0	-126.5	-158.5	-180.6	-91.5	-31.3	-157.0	191.7	139.9	182.6	-38.0	-27.5	-1.4	47.0
	B 2	134.0	-126.1	-78.5	-119.3	-28.2	26.8	-36.6	197.3	134.3	132.7	-17.9	-144.7	-95.1	
	B 3	26.0	-99.7	-140.2	-73.5	-157.5	-163.7	-50.5	153.2	213.5	154.0	-153.2	-178.7	-27.1	
	B 4	-37.0	-367.9	-360.8	-371.4	-134.3	-130.2	-120.1	611.3	200.0	993.9	-346.2	-330.7	-359.6	169.2
	B 5	11.0	-97.0	-115.6	-80.1	-258.6	-186.3	-191.4	136.4	211.9	149.6	-25.4	-17.0	-107.3	
	B 6	88.0	-123.5	-124.9	-57.9	-126.9	12.1	-123.5	143.6	147.8		-70.9	-23.1	-7.3	
	B 7		137.3	124.7	100.4	-115.4	-35.6	-117.6		45.2	146.7	-35.8	-29.4	135.2	
	B 8		117.1	91.9	88.4	-29.5	-15.8	-28.5				-3.1	2.6	37.8	56.1
	B 9														
	B 10														
	BC 1	-122.0	-37.9	-176.8	-170.7	-167.8	-343.9	-194.5			105.3	-137.7	-177.0	-122.4	
	BC 2	162.0	-67.4	-53.9	-102.2	-55.0	-87.0	-167.0	78.2		43.3	81.4	180.0	6.4	56.6
	BC 3	128.0	-112.7	-201.8	-14.4	-167.0	-103.0	-109.5	168.7	122.7	154.2	13.6	5.7	7.0	98.9
	FF 1	32.0	-49.9	-149.6	-88.4										
	FF 2	87.0	-15.6	ROCK	ROCK										
	FF 3	72.0	ROCK	ROCK	ROCK										
	FF 0 m		64.9	69.6	66.2	-123.0	-141.0	-133.0	75.0	74.4	105.4	-66.6	100.4	86.0	
	FF 30 m		89.6	87.8	83.5	-311.0	-67.0	-114.0	117.3	73.3	68.8	140.5	91.3	100.0	
	FF 60 m		ROCK	ROCK	ROCK	-127.0	-78.0	-63.0	73.1	124.0	24.7	-54.3	-46.3	63.6	54.1
Outer Bay North	NB 1	57.0	-105.1	-106.0	-110.3	-148.0	-182.0	-151.6	30.3	85.0	235.4	-38.1	9.3	-115.7	3870.7
	NB 2	-256.0	-35.7	-94.3	-102.5	-388.0	-362.0	-130.0	123.7		178.1	103.1	-68.2	-22.9	63.5
	NB 3	3.0	-55.7	-31.9	-32.8	-172.0	-238.0	-229.0	192.1	183.8	79.8	-310.0	-179.0	-112.8	116.3
	NB 4	63.0	102.6	99.0	102.7	-106.0	-119.0	-153.0	79.8	75.6		-25.4	-78.9	17.8	64.8
	NBC 1	43.0	39.1	11.3	66.5	-262.0	-154.0	-181.0	123.8	73.3	121.4	81.2	-105.1	28.4	
	NBC 2	52.0	100.7	113.7	139.0	-20.0	-2.0	24.0	28.3	81.3	153.9		-77.5	-215.1	
	NBC 3		131.2	124.7	100.4	-137.0	-74.0	-153.0	178.3		21.5	162.9	156.0	-16.2	
Small Bay	SB 1		67.4	108.6	108.7	-143.3	-93.5	-78.7	270.3	268.3	159.0	32.4	-111.1	-100.1	39.8
	SB 2		-122.7	-121.0		-44.2	-36.4	-27.2	211.9	157.4	52.5	-126.8	-6.7	-125.6	40.4
	SB 3		-120.0	-92.6	-29.7	-2.2	-118.7	-60.5	260.6	135.1	210.5	-126.2	-140.6	33.9	123.6
	SBC 1		-122.9	-130.1	-106.0	-52.0	-74.0	-61.2	210.8	160.8	227.5	-124.7	-1.4	-110.6	
	SBC 2		86.7	99.8	96.4	-39.3	-156.3	-96.8	47.6	150.2	256.7	-119.5	-118.3	-135.2	
	SBC 3		-108.2		-124.0	-115.8	-69.4	-40.2	720.6	784.7	304.4	-9.4	-4.7	-2.0	43.7
Small Bay mussel rafts	SBM 1														
	SBM 2														
	SBM 3														
	SBM C1														
	SBM C2														
	SBM C3														

Table 3. Cont. Redox (mV) and Sulphides (µM) measured during the 2019 baseline survey and the 2020-2024 annual chemical survey.

Area	Site	2022		2023						2024					
		Sulph 2	Sulph 3	Redox 1	Redox 2	Redox 3	Sulph 1	Sulph 2	Sulph 3	Redox 1	Redox 2	Redox 3	Sulph 1	Sulph 2	Sulph 3
Big Bay	B 1			-167.0	-20.0	-131.6	<60	<60	<60	-115.6	-72.1	-183.6	124.8	<60	<60
	B 2			-134.0	-141.9	-26.9	<60	<60	<60	-29.8	-157.2	-128.5	<60	<60	119.5
	B 3	126.9		-172.9	-140.0	-35.5	<60	<60	<60	-174.3	-61.8	-136.4	<60	<60	129.5
	B 4	79.9	11601.5	-19.6	12.7	-185.0	180.0	<60	210.0	-463.7	-376.1	-360.7	2938.4	3256.6	2549.2
	B 5			34.7	-217.7	-208.5	<60	<60	<60	-23.8	15.8	-3.4	157.5	153.0	<60
	B 6			-101.9	-125.6	-41.2	<60	<60	<60	-168.1	-78.8	5.1	273.1	952.2	1607.7
	B 7			25.7	-17.1		<60	<60	<60	-155.4	-156.3	-231.5	59.6	<60	<60
	B 8		57.1	13.4	34.9	-20.4	<60	<60	<60	17.9	-53.8	5.6	564.8	<60	55.9
	B 9									-127.4	-110.8	-240.5	126.4	66.0	189.2
	B 10									-95.6	-345.8	-509.3	3355.4	3433.0	3200.6
	BC 1			-217.5	-193.2	-136.0	<60	<60	<60	-286.4	-292.0	-286.0	495.5	388.1	135.7
	BC 2		101.3	-320.9	-315.1	-316.9	<60	<60	270.0	25.5	-217.3	-305.2	881.0	<60	233.6
	BC 3		57.5	-153.3	-171.2	-168.0	<60	<60	<60	-115.7	-21.4	-123.6	398.8	841.7	1021.9
	FF 1			-153.6	-159.4	-43.8	<60	<60	<60	43.5	-25.0	-167.9	118.4	<60	133.1
	FF 2			6.6	-113.6	-73.1	<60	<60	<60	-280.9	-84.6	-340.6	<60	60.4	<60
	FF 3			-109.7	43.9	37.1	<60	<60	<60	-37.4	20.5	28.4	<60	229.9	<60
	FF 0 m			-223.7	-245.9		<60	60.0	<60	2.1	31.9	-14.5	150.1	147.0	794.0
	FF 30 m			-278.5	-108.9	-224.4	<60	240.0	<60	-44.6	30.2	50.2	569.2	132.1	<60
	FF 60 m	66.6		-30.9	15.8	-62.3	<60	<60	<60	-215.2	-252.7	-164.8	44.7	<60	2023.1
Outer Bay North	NB 1		58.0	70.9	-37.4	59.9	<60	<60	50.0	-13.8	-137.1	-130.5	206.2	<60	<60
	NB 2		100.4	-54.0	-247.4	5.6	<60	<60	<60	-14.7	-112.7	-9.9	1530.1	1070.0	<60
	NB 3	66.8		-14.8	-53.3	-78.0	<60	<60	7630.0	-333.2	-221.7		2627.8	2420.9	1437.3
	NB 4			91.7	76.5	-36.3	<60	<60	<60	68.4	54.6	76.4	<60	140.1	<60
	NBC 1			13.1	-221.1	-82.0	<60	<60	<60	38.0	9.3	42.2	60.0	<60	170.0
	NBC 2	118.6		-82.0	-295.7	-5.4	<60	<60	<60	-24.9	-21.6	-117.8	1323.1	1967.5	1644.5
	NBC 3			-5.4	38.8	-141.3	<60	4050.0	<60	81.6	153.9	96.8	<60	<60	<60
Small Bay	SB 1	41.2		-206.8	-263.4	-169.0	<60	<60	<60	-247.8	-198.8		<60	3483.3	387.2
	SB 2	43.3		-188.0	-227.9	-157.6	<60	<60	<60	-116.1	-83.5	-62.9	<60	83.5	<60
	SB 3			-222.4	-220.7	-237.4	<60	<60	<60	-247.4	-372.2	-116.0	<60	288.5	<60
	SBC 1			-225.8	-231.2	-289.9	<60	<60	80.0	-84.0	-171.6	-88.2	<60	<60	<60
	SBC 2			-218.3	-258.8	-164.4	<60	<60	<60	-263.4	-92.7	-14.6	<60	91.7	<60
	SBC 3	94.1	79.8	-160.1	-304.8	-144.9	80.0	<60	<60	-166.7	-135.5	-47.5	311.6	2498.3	315.3
Small Bay mussel rafts	SBM 1									-417.6	-410.1	-373.1	25492.3	26690.2	1249.2
	SBM 2									-374.1	-359.6	-366.7	545.5	13169.9	14040.3
	SBM 3									23.8	-57.4	-133.6	135.3	124.8	277.2
	SBM C1									-12.3	-111.8	-4.8	<60	93.9	<60
	SBM C2									-112.3	-156.8		<60	102.8	<60
	SBM C3									-109.3	-149.8	-109.2	175.4	<60	<60

7 APPENDIX 2



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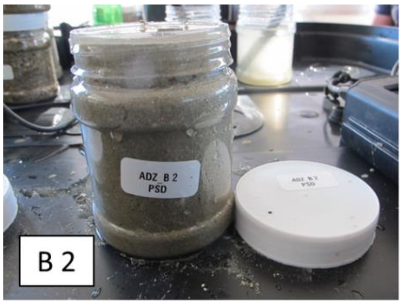
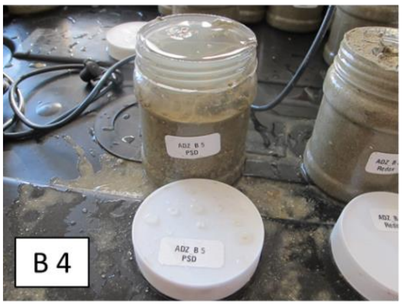


B 2

B 3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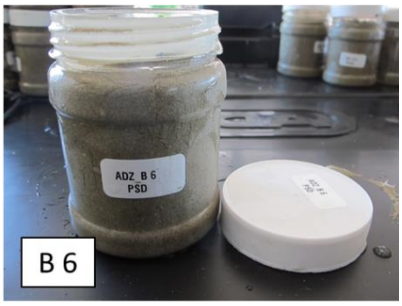

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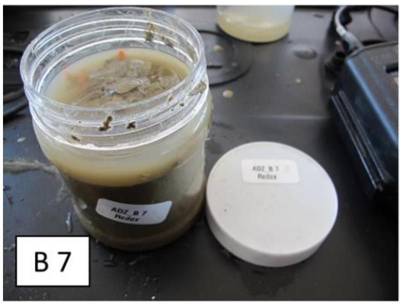
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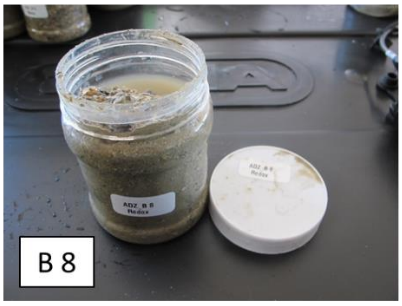
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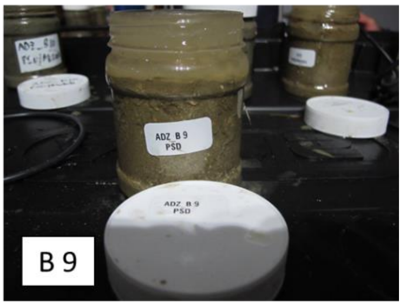

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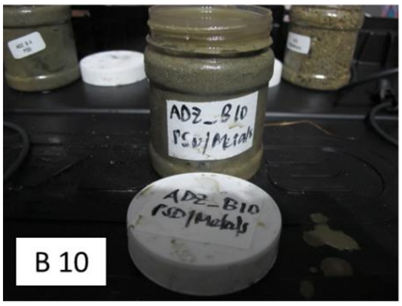

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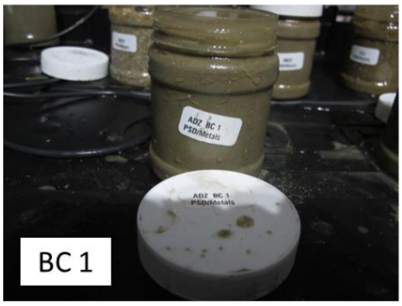
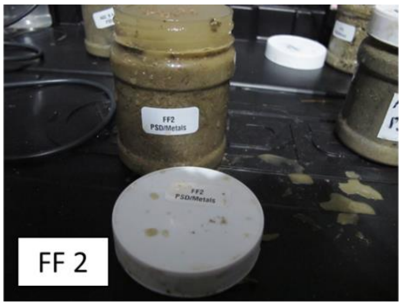

BC 1


BC 2

BC 3

FF 1



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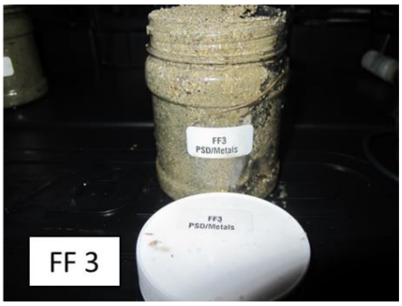

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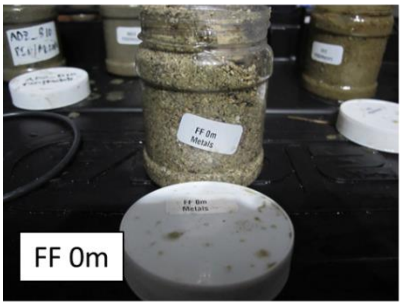

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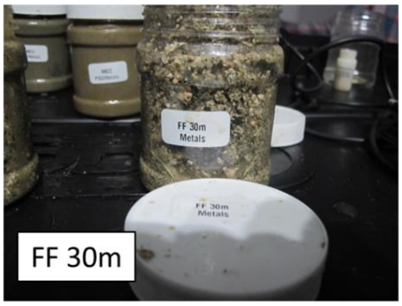
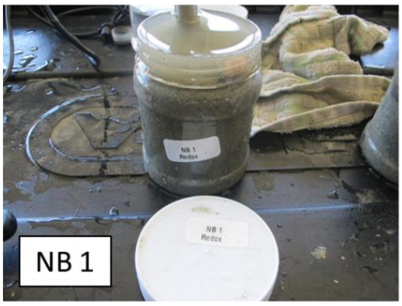


FF 30m

FF 60m



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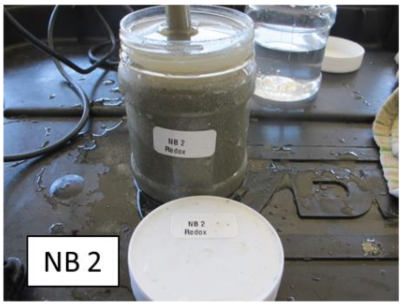

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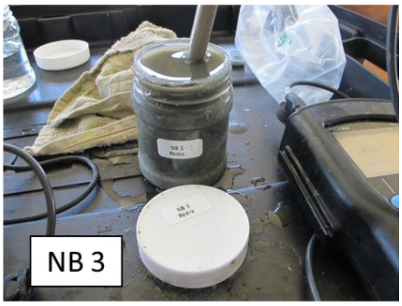
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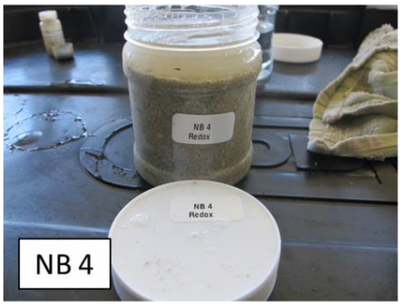

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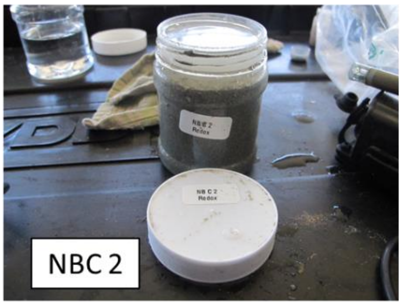
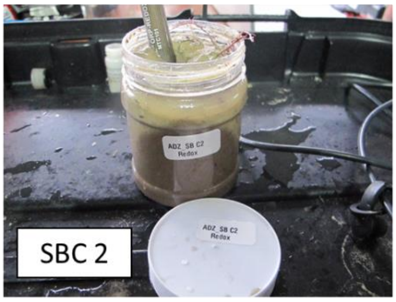

NBC 2


NBC 3

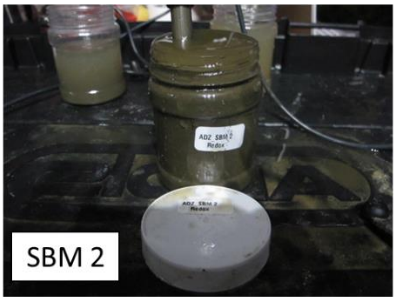

SB 1

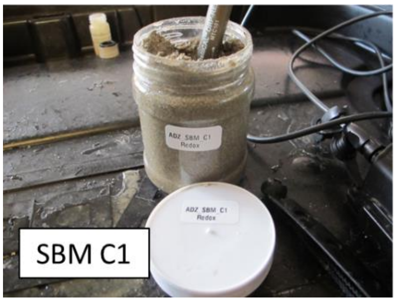
SB 2

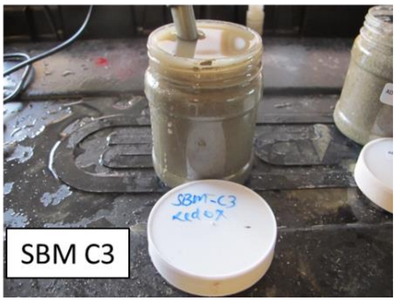
SB 3

SBC 1

SBC 2

SBC 3

SBM 1

SBM 2

SBM 3

SBM C1

SBM C2

SBM C3

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