Coastal Systems Research Group
Ecosystem Services Competence Area

Draft Report

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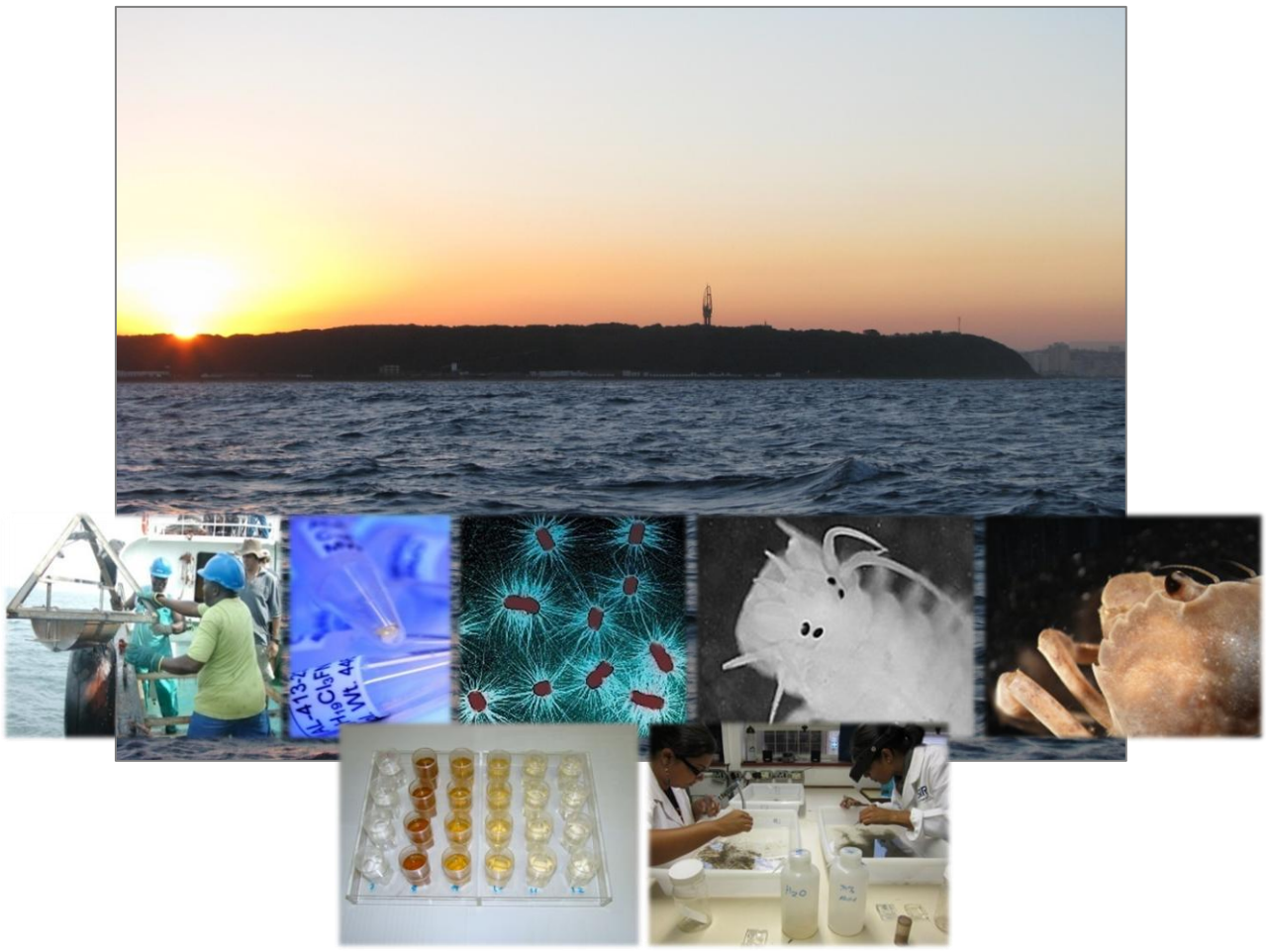
ETHEKWINI MUNICIPALITY

SEA DISPOSAL OF SEWAGE: ENVIRONMENTAL SURVEYS IN THE DURBAN OUTFALLS REGION

Report no 30

Surveys made in 2011



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Obtaining Copies of Reports

Copies of this technical report and a companion non-technical summary report are available in printed and electronic formats. Should you wish to obtain copies of the reports please contact Brent Newman of the CSIR at 031 - 242 2377 or bnewman@csir.co.za.

Executive Summary

In common with large coastal cities in many regions of the world, a significant proportion of the wastewater that is generated daily by households and industries in the city of Durban is discharged to the marine environment through deepwater outfalls. The bulk of the wastewater is discharged through outfalls that serve the Central Works and the Southern Works wastewater treatment facilities, both owned and operated by the eThekwini Municipality. The outfalls are situated about 12 km apart off the Bluff/Merewent area of Durban, and discharge effluent at a depth of about 43 - 53 m in the case of the Central Works outfall (about 3.2 km off the shoreline) and about 54 - 64 m in the case of the Southern Works outfall (about 4.2 km off the shoreline). The Central Works outfall carries predominantly sanitary effluent, that is, wastewater from the kitchens, bathrooms and toilets in homes and workplaces. The Southern Works outfall carries both sanitary and industrial wastewater, that is, also wastewater from manufacturing related processes.

These effluents contain a range of contaminants and biological material that have the potential to impair the ecological functioning of the receiving water and to compromise human health. To ensure that the integrity of the receiving water is not unacceptably compromised South Africa's national government issues effluent discharge licenses. Discharge licenses stipulate the conditions under which the discharge is authorised. One of the conditions is that the effluent discharger must implement and report on an environmental monitoring programme designed to determine the (potential) impact of the discharge on the receiving water. This report analyses and summarises the findings of a survey performed in May 2011 to identify the potential environmental impact of effluent discharged through the Central Works and Southern Works outfalls on the receiving marine environment. A purpose of this report is to demonstrate compliance with conditions of authorisation for effluent discharge in terms of environmental impact monitoring and reporting. The report serves the equally important roles of providing wastewater managers from the eThekwini Municipality with strategic information for managing the discharges and of informing the public on the status of the receiving marine environment.

The overall objective of the Durban outfalls monitoring programme is to identify impacts (with the main focus on adverse impacts) to the receiving marine environment that can be attributed to effluent discharge through the Central Works and Southern Works outfalls. The monitoring programme uses various indicators - physical, chemical and biological indicators - to reach a conclusion on impacts. The indicators were identified because they provide information on the condition of the marine receiving environment and/or because they are tracers (or signals) of effluent. Examples of indicators include the presence of pathogenic bacteria in water and metal concentrations in sediment. The information obtained from the different indicators is then used in a weight of evidence approach for reaching conclusions on impacts. In other words, conclusions are reached based on the information of many lines of evidence, rather than a single line of evidence. This is a more reliable approach for identifying impact in the inherently variable and 'noisy' natural world. The monitoring programme comprises several major foci, each of which is presented as a separate chapter in this report. The foci include effluent toxicity, water and sediment quality, and the status of benthic macrofaunal communities. Additional chapters provide a general introduction to the monitoring programme, including a discussion of the survey sampling design, and a synthesis of the key findings and the way forward.

Effluent Toxicity

Although the concentrations of chemicals measured in effluent can be used to estimate its potential toxicity, chemical analyses alone are inadequate for regulating effluent discharges for several reasons. It is impossible to measure for the estimated 10 000 chemical substances that are in regular use in the world, and the vast majority of these chemicals have no associated toxicity information. Furthermore, many chemicals with known toxicity have unknown additive, antagonistic or synergistic toxic effects when present in complex mixtures, such as in effluent. Effluent that is thoroughly characterised

chemically and considered safe on this basis can still be toxic due to the presence of unknown constituents and unknown interactive effects of the constituents. It is for this reason that environmental regulatory authorities in many regions of the world require the routine toxicity testing of effluent. In the context of effluent testing, the tests are known as whole effluent toxicity tests because the tests measure the combined effects of all chemical constituents in the effluent. The test that is used to measure the toxicity of effluent discharged through the Central Works and Southern Works outfalls is the sea urchin fertilisation test. A key measure of the test is the minimum acceptable toxicant dilution, which defines the minimum number of dilutions in seawater that must be achieved to render the effluent non-toxic to sea urchin gametes. Armed with knowledge of the effluent's minimum acceptable toxicant dilution and the theoretical minimum initial dilution of the outfalls, it is possible to define an area around the outfall where a toxic impact might be expected.

On none of the 12 testing dates did the number of dilutions required to render final effluent from the Southern Works wastewater treatment facility non-toxic to sea urchin gametes exceed the lowest theoretical minimum initial dilution of the outfall that serves this facility. This indicates that there was little risk of toxicity beyond the zone of initial dilution for the outfall. The situation was different for the final effluent from the Central Works wastewater treatment facility. Final effluent could not be collected on four testing dates, because of problems experienced at the facility. On three of the eight dates that final effluent was tested, the number of dilutions required to render the effluent non-toxic to sea urchin gametes exceeded the lowest theoretical minimum initial dilution of the outfall that serves this facility. Thus, when there is no current flow (i.e. when the lowest theoretical minimum dilution occurs), similarly sensitive organisms beyond the zone of initial dilution in the marine receiving environment may have experienced toxic effects. The probability that toxic effects manifested is probably low considering that it is highly improbable that the receiving marine environment is ever stagnant and the number of dilutions required to render final effluent from the Central Works wastewater treatment facility non-toxic to sea urchin gametes was not much higher than the lowest theoretical minimum initial dilution of the outfall.

The toxicity of final effluent from Central Works wastewater treatment facility was often higher than the toxicity of final effluent from the Southern Works wastewater treatment facility. This was contrary to expectation considering that the Southern Works wastewater treatment facility receives a high volume of industrial effluent, which was expected to reveal in a higher toxicity. Problems experienced at the Central Works wastewater treatment facility may be a reason for the generally higher and more variable toxicity recorded for final effluent from this facility.

Water Quality

Water quality monitoring in the proximity of outfalls serves two main purposes. First, measurements can be used to track the dispersion and dilution of effluent and thereby assess whether the outfall is meeting system design specifications. Second, measurements can be used to assess potential risks posed to organisms in receiving waters inside and outside of the zone of initial dilution, by comparing measured variables to water quality guidelines for the protection of aquatic ecosystem health. Regulatory authorities in many countries typically require that measured physico-chemical variables meet water quality targets at the boundary of the zone of initial dilution. The principal focus of the water column physico-chemistry component of the monitoring programme is to determine whether water quality at the margin of the zone of initial dilution at the time of monitoring complies with South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment).

Discrete water samples were collected from the middle and bottom of the water column at three stations situated to the immediate north-northeast and south-southwest of the diffuser sections of both the Central Works and Southern Works outfalls, at the margin of the zone of initial dilution (i.e. a total of 18 samples per outfall). The margin of the zone of initial dilution was calculated as three times the average water column depth for the diffuser section (i.e. 144 m for Central Works outfall and 177 m for Southern Works outfall). Discrete surface water

samples were also collected at each of nine reference sites. Temperature, salinity, pH, dissolved oxygen, turbidity and chlorophyll-*a* concentrations were profiled *in situ* at stations near the outfalls and at reference sites.

Various physical, chemical and biological variables and responses were measured *in-situ* and in discrete water samples collected on a single occasion in May 2011. The data do not, therefore, provide a synoptic understanding of the variability of the variables and responses in the study area but only at the time of monitoring.

Of the various physical, chemical and biological variables measured *in-situ* at the margin of the zone of initial dilution for the Central Works and Southern Works outfalls, none showed anomalies that could confidently be attributed to effluent discharge. Faecal indicator bacteria counts provided the clearest effluent signal. None of the other indicators measured provided signals that could confidently be attributed to effluent discharge. None of the water samples was toxic to sea urchin gametes.

The values and concentrations of the majority of physical and chemical variables were compliant with the South African Water Quality Guidelines for Coastal Marine Waters. There were non-compliances for pH, dissolved oxygen concentrations and copper. For pH, the non-compliances were spurious because the upper limit for this guideline is too low and is routinely exceeded in nearshore marine waters off the KwaZulu-Natal coast. For dissolved oxygen, the non-compliance is linked to a natural oceanic phenomenon (stratification) rather than effluent discharge. The dissolved oxygen concentrations were also not so low as to be of ecological concern. Two copper concentrations were non-compliant with the relevant guideline. One of these concentrations was in fact very high. However, it is difficult to link these concentrations to effluent discharge, since they were detected in samples collected a substantial distance from the outfalls, and concentrations at most stations were below the method detection limit.

Trends for most physical and chemical variables were qualitatively similar to those for the 2009 and 2010 surveys of the Durban outfalls monitoring programme.

Sediment Quality

The major focus of the Durban outfalls monitoring programme is on the benthic environment. The reason is that sediment is the predominant sink for many contaminants that are anthropogenically introduced in solution to saline surface waters. Many contaminants have a low water solubility and are particle reactive, and once introduced in solution to (especially marine) surface waters rapidly adsorb onto suspended sediment and organic matter and are in this manner 'scavenged' from the water column through flocculation, coagulation and sedimentation. Under suitable conditions, contaminants may accumulate in sediment to such high concentrations that these adversely affect benthic communities. In addition to these environmental concerns there are several pragmatic reasons for focusing more attention on contaminants in sediment than in the water column. The low and often highly variable concentrations of contaminants in the water column due to variations in turbulence and mixing and variable anthropogenic inputs mean that only a snapshot of water quality status and trends is gained. Important contamination events may be missed. Analysis of sediment provides a far more conservative, spatially and temporally integrated measure of contamination problems. Furthermore, contaminant concentrations in sediment are usually orders of magnitude higher than in the overlying water column (up to a million times more) and this makes detection and measurement in the laboratory easier.

The most significant impacts of effluent discharge through deepwater marine outfalls are commonly evident in the benthic compartment of the receiving environment. The Durban outfalls monitoring programme places particular focus on this compartment of the receiving environment for several reasons. In contrast to the water column, where high temporal and spatial variability due to turbulence and mixing means that monitoring provides only a snapshot of conditions at the time of monitoring and contamination events may be missed, the benthic environment provides a far more conservative, spatially and temporally integrated measure of conditions. Conditions in sediment are more stable than the water column and because most contaminants are particle reactive they tend to

accumulate in sediment rather than remain in solution.

The findings of the 2011 survey of the Durban outfalls monitoring provide clear evidence that the discharge of effluent has impaired sediment quality in the vicinity of the diffuser sections of the Central Works and Southern Works outfalls. The impacts were, however, more frequent and of a greater spatial extent and severity in the vicinity of the Southern Works outfall. Sediment near both outfalls was characterised by high faecal indicator bacteria colony forming unit counts. In fact, faecal indicator bacteria were detected at all sites, including the reference sites, providing evidence that effluent was impinging on the benthic environment across the study area.

Sediment near the Southern Works outfall and to a far lesser degree and extent at the Central Works outfall was enriched with particulate organic matter. This has presumably caused the higher chemical oxygen demand of sediment near both outfalls as compared to reference sites, although once again the effects were more pronounced near the Southern Works outfall. At the Southern Works outfall, the accumulation of organic matter and the associated chemical and probably also biological oxygen demand clearly exceeded the rate of re-ventilation of the sediment with dissolved oxygen. This is evident in the strong aroma of hydrogen sulphide and discolouration of the sediment. Hydrogen sulphide is produced by heterotrophic bacteria under anoxic conditions, while the discolouration of sediment is probably due to sulphide binding ionic metals. Metal-sulphide complexes usually impart a dark brown or black colour to sediment.

Grain size normalised concentrations of several metals in sediment near both outfalls were higher than for sediment from most of the reference sites and provides evidence that the sediment was metal contaminated. Of the wide suite of organic chemicals analysed, the use of many of which is regulated under the Stockholm Convention on Persistent Organic Pollutants, only polycyclic aromatic hydrocarbons were measured at concentrations exceeding the method detection limit, and then only at three sites in the immediate vicinities of the outfalls and at one reference site. Although there was evidence for the contamination of sediment in the immediate vicinities of the outfalls by metals and polycyclic aromatic hydrocarbons, comparison of metal and polycyclic aromatic hydrocarbon concentrations to sediment quality guidelines suggests there is theoretically little probability that these contaminants were adversely impacting benthic macrofaunal communities.

However, sediment porewater at the majority of outfall and reference sites was toxic to sea urchin gametes. The magnitude of toxicity at Central Works outfall and reference sites was low with the exception of a reference site situated 6000 m to the southwest of the outfall. The toxicity of sediment porewater at Southern Works outfall reference sites was comparable and relatively mild, but was high at sites situated in the immediate vicinity of the outfall. The cause of the toxicity could not be satisfactorily attributed to ammonia concentrations measured in porewater, nor to concentrations of metals and polycyclic aromatic hydrocarbons in sediment. There is a possibility that the high toxicity of porewater at some sites in the immediate vicinity of the Southern Works outfall may be due to the presence of hydrogen sulphide, which is highly toxic to most marine organisms.

The most significant impacts of effluent discharge on the chemical properties of sediment occurred within about 300 m of the diffuser sections of the outfalls, although some impacts did extend as far as 500 m of the outfalls.

There was little difference in trends for most of the indicators of environmental condition between the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. In other words, the impact of effluent discharge from the Central Works and Southern Works outfalls on the benthic environment in the study area has remained broadly comparable for the latter period.

Benthic Macrofauna

A critical end-point of any outfall monitoring programme is to determine whether effluent discharge is adversely impacting the ecology of the receiving environment. Little relevance can be attached to a particular degree of contamination of

water and sediment unless it can be evaluated within a biological context. Whereas sediment chemistry can only provide a screening level assessment of potential effects, the monitoring of biological communities provides a direct measure of effects. The Durban outfalls monitoring programme places considerable emphasis on an analysis of the structure and composition of benthic macrofaunal communities as an indicator for measuring the ecological impact of effluent discharge. Benthic macrofauna are invertebrate organisms greater than 1 mm in size that live in or on the sediment.

The structure of marine benthic macrofaunal communities is influenced by many factors. These include abiotic factors, such as the sediment conditions, salinity and temperature, as well as biotic factors such as food availability, competition and predation. A major challenge in environmental monitoring is to distinguish between naturally occurring and anthropogenically induced changes to benthic macrofaunal communities. This is best achieved through comparison of communities from impacted sites to those from reference sites. While benthic community data has limitations, properly analysed they remain the most ecologically relevant line of evidence regarding possible impacts on the benthos.

Univariate and multivariate analysis of benthic macrofaunal community structure for the 2011 survey of the Durban outfalls monitoring programme provides clear evidence that the seabed near the Southern Works outfall is enriched with particulate organic material. Benthic macrofaunal community structure in close proximity to the outfall has been modified because of this enrichment. This is manifested by reduced biodiversity and an increased abundance of opportunist capitellid polychaetes. Comparison with earlier surveys reveals a gradual increase of this effect over the past decade. While this impact is not considered to pose an immediate ecological threat, its expansion is cause for concern and should be accounted for in management considerations.

The Central Works outfall appears to be operating within the assimilative capacity of the receiving environment. In 2011, however, some indication was given of impacts manifesting in benthic macrofauna in close proximity to the outfall. Benthic macrofaunal community response at affected stations appears to be similar to that at impacted stations near the Southern Works outfall and is likely the result of mild organic enrichment of the sediment. This is presently of little concern but requires close monitoring in future surveys.

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Chapter 1
General Introduction

1.1. Introduction

In common with large coastal cities in many regions of the world, a significant proportion of the wastewater that is generated daily by households and industry in the city of Durban is discharged to the marine environment through deepwater outfalls. The bulk of the wastewater is discharged through outfalls that serve the Central Works and the Southern Works wastewater treatment facilities, which are owned and operated by the eThekwini Municipality.

Effluent contains a range of contaminants and biological material that has the potential to impair the ecological functioning of the receiving water and to compromise human health (e.g. Stevens et al. 2003, Nakada et al. 2004, Moon et al. 2008). Based on estimates of environmental releases, wastewater is regarded as a leading source of anthropogenically derived contaminants to coastal waters (e.g. Chambers et al. 1997). These contaminants are generally described within categories such as solids, substances that exert a biological and chemical oxygen demand, nutrients, pathogens, organic chemicals, metals, and soaps, oils and grease. More recently, concern has shifted to so-called emerging contaminants, which are chemicals included in personal care and pharmaceutical products. Since most wastewater treatment facilities only perform primary treatment to decrease the load of suspended solids, nutrients and floatables, there is an inefficient removal of other contaminants that are consequently introduced to the receiving water (Chambers et al. 1997). Nevertheless, the managed discharge of treated wastewater (effluent) to the marine environment is recognised as an acceptable disposal option from multiple perspectives, including human and environmental health, social acceptability, and economic prudence.

Whether the discharge of effluent impairs the ecological functioning of a receiving water depends on the receiving waters assimilative capacity, that is, its capacity to receive effluent or toxic materials without deleterious effects to aquatic life or humans who consume and/or otherwise use the water. The assimilative capacity can be considered as the receiving waters 'pollution diet' - too much pollutant loading and deleterious effects will manifest. Assimilative capacity differs between receiving waters depending on the nature of the effluent and the characteristics of the receiving water in terms of its ability to dilute, disperse and degrade contaminants. Not surprisingly, the voluminous and high-energy marine environment has a higher assimilative capacity compared to smaller volume sheltered waters, such as estuaries. Of importance is the volume of effluent discharged. Thus, while the absolute concentrations of contaminants in effluent might be low and elicit no acute toxic effects, the persistent introduction of contaminants may overwhelm the assimilative capacity of the receiving water in the long-term and lead to chronic toxicity.
To ensure that the integrity of the effluent receiving waters is not unacceptably compromised, South Africa's national government issues effluent discharge licenses[1]. The licenses stipulate the conditions under which the discharge is authorised. One of the conditions is that the effluent discharger must implement and report on an environmental monitoring programme that is designed to determine the impact of the discharge on the ecology of the receiving water and on human health.

This report analyses and discusses the findings of a survey performed in May 2011 to identify the impact of effluent discharged through the outfalls that serve the Central Works and Southern Works wastewater treatment facilities on the ecology of the marine receiving environment. Important to note is that this report does not focus in detail on the human health impacts. Such impacts are addressed in a different monitoring programme conducted at recreational

[1] In the past, the Department of Water Affairs was the government agency mandated with the control of wastewater discharges to all surface waters of South Africa in terms of the National Water Act (Act 36 of 1998) (NWA). With the promulgation of the National Environmental Management: Integrated Coastal Management Act (Act 24 of 2008) (ICM Act), the responsibility of regulating effluent discharges to the marine environment transferred to the Department of Environmental Affairs. Therefore, in future licensing of marine effluent discharges will become the responsibility of DEA under the ICM Act. However, the issuing of effluent discharge licenses for freshwater receiving waters remains the responsibility of the Department of Water Affairs under the NWA.

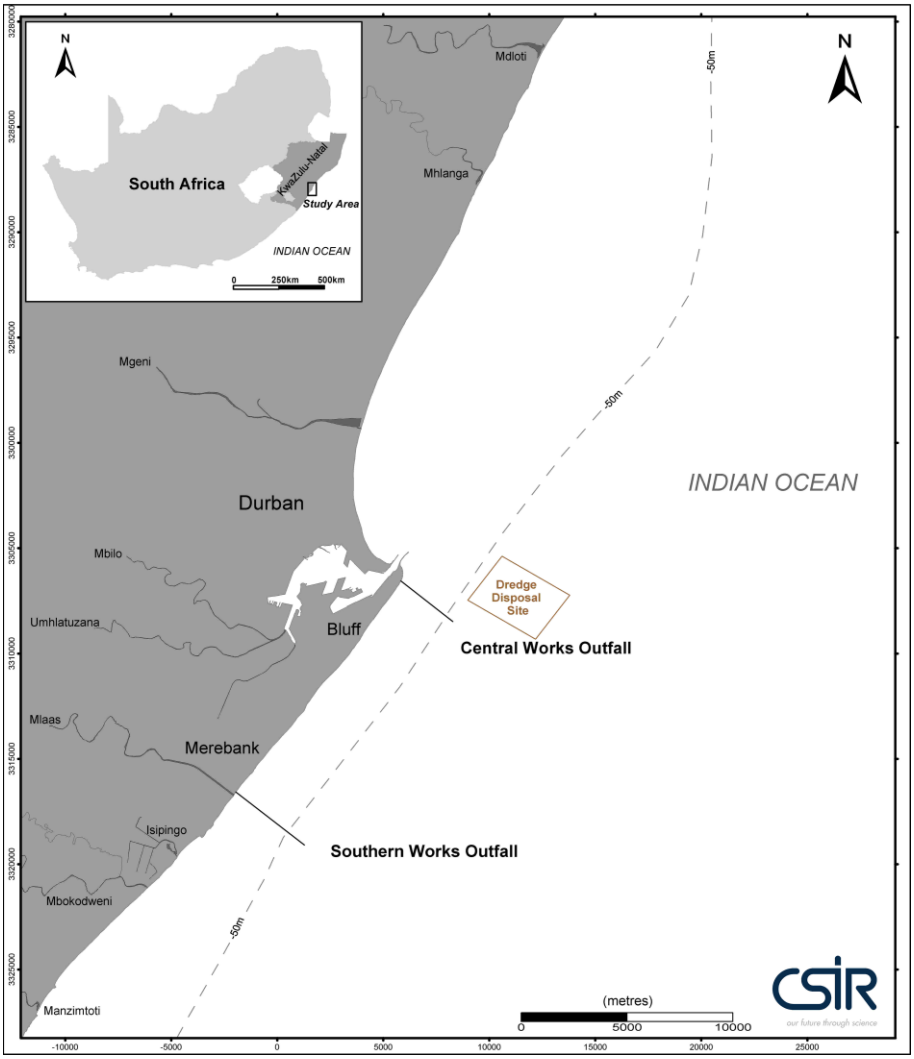

Figure 1.1. Map illustrating the positions of the Central Works and Southern Works outfalls. Note that the 50 m bathymetric line is approximate.

beaches along Durban's shoreline by the municipality.

One purpose of this report is to demonstrate compliance with conditions of authorisation for effluent discharge in terms of environmental impact monitoring and reporting relating to ecological health. The report serves the equally important roles of providing managers of the Central Works and Southern Works wastewater treatment facilities with strategic information for managing the discharges and of informing the public on the status (or health) of the receiving water.

Although this report deals with technical issues, where possible these are discussed and presented in a simplified form for the benefit of non-specialist audiences. A glossary of terms has also been included for this purpose, and wherever possible the use of acronyms is avoided. A non-technical summary of the key findings of the 2011 survey is also available. Copies of technical and non-technical reports are freely available to the public in printed and electronic formats (see procedure for obtaining reports on page ii).

1.2. Brief Description of the Receiving Marine Environment

The predominant oceanographic feature off the KwaZulu-Natal coastline is the Agulhas Current, a strong-flowing western boundary current with core speeds as high as 2 $m.s^{-1}$. Since the continental shelf (defined as <200 m isobath) is narrow (<11 km) and has a steep gradient along most of the KwaZulu-Natal coastline, the core of the current flows close inshore, following the shelf edge, and generally has a strong influence on nearshore oceanography (Tripp 1967, Schumann 1988, Lutjeharms et al. 2000). Water column depth tends to increase rapidly with distance offshore, to the extent that off the Bluff area of Durban (see Figure 1.1) the 50 m contour is

only about 3 - 4 km offshore.

Current flows close inshore off the Bluff show a dominant northward component (Schumann 1982, CSIR 2008), but with increasing distance offshore the current is about equally divided between southward and northward flow (Harris 1964, Pearce et al. 1978) until in the Agulhas Current itself the flow is predominantly southward (Schumann 1982, Schumann 1988). Currents close inshore often flow strongly, reaching up to 1 $m.s^{-1}$ through the water column. North-northeast flows are generally stronger than south-southwest flows (CSIR 2008). As a consequence of current flow the predominant direction of sediment bedload transport close inshore off the Bluff is northward (Flemming and Hay 1988).

Coastal winds in the region blow mainly along an axis that is parallel to the coast, that is, north-northeasterly and south-southwesterly (Schumann 1989). Average wind speeds are about 2.5 $m.s^{-1}$ and exert a strong influence on the movement of surface waters. Surface seawater temperatures close inshore are about 24 to 25^{o}C in summer and 20 to 21^{o}C in winter. Thermal stratification of the water column is typically weak (Pearce 1978, Schumann 1998).

1.3. Outfall Design and Nature of the Effluent

Prior to the commissioning of the Central Works and Southern Works outfalls a large proportion of the wastewater generated by households and industry in the city of Durban was discharged into the entrance channel to Durban Bay, on the outgoing tide. Effluent was also discharged into the surfzone at Fynnlands, on the Bluff. The consequence of these practices was gross pollution of shoreline waters, especially by faecal bacteria (Livingstone 1969), leading to a significant public health concern. In 1959, the Natal Provincial Administration, through its Town and Regional Planning Commission, established the Marine Disposal of Effluents Fellowship. This body examined all aspects of point source effluent discharge into the marine environment off KwaZulu-Natal and provided a theoretical framework for the siting and construction of deepwater outfalls. In 1964, the erstwhile Durban Corporation launched an intensive investigation into the feasibility of constructing deepwater outfalls for the efficient and safe discharge of effluent into the marine environment. Design calculations were based on extensive oceanographic measurements. It was decided that two outfalls, located about 12 km apart, would be constructed, one serving the central city catchment and situated towards the northern extremity of the Bluff (Central Works outfall, Figure 1.1), the other serving the southern city catchment and situated near the mouth of the Mlaas Canal (Southern Works outfall, Figure 1.1). The Southern Works outfall was commissioned in November 1968 and the Central Works outfall in November 1969, and have since been operating continuously. The outfalls are constructed of steel encased in concrete and have the dimensions provided in Table 1.1.

The Central Works outfall carries predominantly sanitary wastewater (i.e. that which is flushed down toilets or rinsed into drains in houses and commercial facilities), while the Southern Works outfall carries both sanitary and industrial wastewater (i.e. also wastewater derived from manufacturing related processes).

Numerical modelling of the dilution of effluent discharged through the Central Works and Southern works outfalls by the CSIR (CSIR 1998) demonstrated that the theoretical minimal initial dilution for each outfall varies depending on the season (Table 1.2). To understand the meaning of the term minimum initial dilution it is necessary to consider the behaviour of effluent after it is discharged through an outfall situated on the seabed. Effluent is discharged through a series of ports situated along the diffuser section of each outfall (the terminus of each outfall is capped). The diffuser ports are hydraulically designed to create a turbulent ejection jet, which serves to intensely mix effluent with the receiving water. This results in a rapid and large reduction in the concentration of effluent constituents. Further mixing (and hence dilution) occurs when the buoyant effluent ascends through the water column until it reaches neutral buoyancy, at which point the effluent spreads laterally, creating a horizontal dispersing layer. The effluent is buoyant because it is essentially comprised of freshwater, which is less dense than seawater and, therefore, ascends through the water column until it reaches neutral buoyancy. The area over which effluent is

dispersed until such time that it reaches neutral buoyancy is defined as the zone of initial dilution (sometimes called the nearfield mixing zone).

The zone of initial dilution is important because the rate of dilution and associated changes in water quality within this zone are typically high, due to discharge momentum and buoyancy effects, but usually decreases considerably outside of this zone (often referred to as the secondary dilution zone or the farfield mixing zone), where dilution is influenced by wind, wave and current induced turbulence.

The minimum initial dilution achievable for an outfall is a theoretical calculation that refers to the minimum number of times that effluent will be diluted by the receiving water in the worst case scenario, that is, when there is zero current flow.

As stated previously, numerical modelling has demonstrated that the theoretical minimal initial dilution for the Central Works and Southern Works outfalls varies seasonally. In summer, the effluent plume is often trapped (well) below the surface due to thermal stratification of the water column, that is, when the water column is characterised by a sharp decrease in temperature over a relative narrow depth range. Since waters of different temperature have a different density, the density related differences either side of the thermocline impede mixing and trap effluent. This trapping of effluent reduces the number of initial dilutions achievable, since there is less time for mixing during the buoyant rise of the effluent before it disperses horizontally. In winter, thermal stratification of the water column is less pronounced and results in large rise heights and a corresponding increase in the minimum initial dilution achieved. In fact, numerical modelling predicts that the effluent plume will reach the surface at a frequency of once every two days in winter, but never in summer. The prediction for spring falls between the predictions for summer and winter.

The reader will appreciate that it is seldom, and probably never that the theoretical minimum initial dilution will be realised in the marine environment for the reason that the water column is always moving. However, following a precautionary approach the theoretical minimum initial dilution is typically used to assess potential environmental impacts. Of interest in this context is the maximum initial dilution that may be achieved for the Central Works and Southern Works outfalls, which numerical modelling demonstrates may be as high as 195 044 and 136 015 times respectively.

Table 1.1. Design dimensions of the Central Works and Southern Works outfalls.

Feature	Central Works	Southern Works
Length of outfall from shoreline (m)	3 200	4 200
Main diameter (m)	1.32	1.37
Length of diffuser section (m)	450	290
Number of diffusers	18	34
Depth of diffuser section (m)	43 - 53	54 - 64
Capacity (megalitres per day)	135	230

1.4. Components of the Monitoring Programme

The impact of effluent discharge on a receiving water can be assessed through the measurement of physical and chemical variables that provide a tracer (or signal) of the effluent in water, sediment and

Table 1.2. Numerical model predictions of the minimum initial dilution, effluent rise height and incidence of effluent surfacing for the Central Works and Southern Works outfalls (from CSIR 1998).

		Initial Dilution			Rise height (m)			Plume Surfacing
Outfall	Season	Minimum	Median	Maximum	Minimum	Median	Maximum	(%)
Central Works	Summer	229	671	5 336	8.0	17.5	38.6	0
	Winter	375	4 429	178 330	11	51.1	55.4	53
	Spring	317	934	195 044	7.8	16.3	55.4	6
	Annual	229	947	195 044	7.8	20.8	55.4	20
Southern Works	Summer	272	654	90 466	8.3	19.2	57.6	0
	Winter	424	2 871	95 168	13.7	47.6	57.6	48
	Spring	261	931	136 015	9.0	15.7	57.6	9
	Annual	261	975	136 015	8.3	21.5	57.6	19

biological tissue, and/or through the assessment of biological communities that characterise the ecological condition of their habitat. Various physical, chemical and biological indicators of environmental condition are measured in the Durban outfalls monitoring programme and are then used in a weight-of-evidence approach to reach a conclusion on the impact of the discharges. A brief rationale and frame of reference for the different components of the monitoring programme and associated indicators is provided below.

The Durban outfalls programmes is one of the longest, continuous monitoring programmes in the world and, in a South African context, is undoubtedly the most comprehensive outfall monitoring programme.

1.4.1. Effluent Toxicity

Although the physical and chemical characteristics of effluent can be used to estimate its potential toxicity, this information alone is inadequate for regulating effluent toxicity for several reasons. First, it is impossible to monitor the estimated 10 000 chemical substances that are in regular use in the world. It is generally recognised that in complex effluent samples only a small fraction (around 20%) of the substances present can be identified. Second, many of the chemicals that can be detected have little or no toxicity information available. Third, many chemicals with known toxicity have unknown additive or synergistic toxic effects when present in complex mixtures. Consequently, effluents that are characterised physically and chemically and are considered safe on this basis can still be toxic due to unknown constituents and unknown interactive effects of the constituents. It is for this reason that regulatory authorities in many regions of the world require the routine toxicity testing of effluent. The tests are known as whole effluent toxicity tests because they measure the combined effects of all physical and chemical constituents in the effluent.

Whole effluent toxicity testing entails exposing test organisms (e.g. fish, larval invertebrates, algae) to serial dilutions of effluent for a predefined period. A dose-response curve can then be obtained by evaluating adverse effects, such as mortality, at the different dilutions. Scoring criteria (e.g. the lethal concentration for 50% of organisms; LC_{50}) can then be determined from the dose-response curve through various statistical procedures.

Assessing the toxicity of whole effluent to the gametes of sea urchins is an important component of the Durban outfalls monitoring programme. Sea urchin gametes are amongst the most sensitive life stages for evaluating the toxicity of effluent (e.g. Woodworth et al. 1999). Although no attempt is made to correlate toxicity to specific constituents of the effluent, results of toxicity tests provide a measure for assessing temporal differences in effluent toxicity and for identifying the potential for adverse impacts in the receiving water. A key scoring criteria is the Minimum Acceptable Toxicant Dilution, which is the minimum number of dilutions in seawater required to render the effluent non-toxic to sea urchin gametes. Armed with a knowledge of the effluent's Minimum Acceptable Toxicant Dilution and the theoretical minimum initial dilution of the outfalls, it is possible to determine whether toxic effects to similarly sensitive organisms/life stages is possible beyond the margin of the zone of initial dilution.

It is impractical to continuously measure the toxicity of effluent from wastewater treatment facilities. Therefore, the Durban outfalls monitoring programme focuses on measuring the toxicity of a random sample of final effluent discharged from the Central Works and Southern Works wastewater treatment facilities at monthly intervals.

1.4.2. Water Quality

The main objective of the water quality component of the Durban outfalls monitoring programme is to determine whether water quality at the margin of the zone of initial dilution at the time of monitoring was compliant with the South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995) and compares favourably with reference sites. For this purpose, a suite of physical, chemical and biological variables are monitored at the margin of the zone of initial dilution, and for comparative purposes also at numerous reference sites.

Since water quality monitoring is performed on a single occasion per annum its value has been questioned. The reason is that the physical, chemical

and biological properties of the water column are highly variable, due to variable turbulence and mixing. Furthermore, episodic large-scale natural events in the receiving water that are poorly understood make it difficult to determine whether anomalies in the data are due to effluent discharge or simply reflect the poor understanding of the influence of these natural events on water physico-chemistry.

Regulatory authorities in many countries have dispensed with the need to monitor the concentrations of contaminants in the water column near effluent discharges, because of the large degree of variability in the water column. Rather, water column monitoring usually focuses on *in situ* measurements of so-called conventional variables, with the principle objective being to locate the effluent plume and on the measurement of faecal indicator bacteria to determine whether the effluent is impinging on the shoreline and posing a public health risk. Where water column monitoring is performed then it is often performed at a high frequency (monthly), so that the frequency of water quality guideline non-compliance can be determined.

1.4.3. Sediment Quality

The main focus of the Durban outfalls monitoring programme is on the benthic environment. The reason is that sediment is the predominant sink for many contaminants anthropogenically introduced in solution to surface waters. Many contaminants have a low water solubility and are particle reactive, and once introduced in solution to (especially marine) surface waters rapidly adsorb onto suspended sediment and organic matter and are in this manner 'scavenged' from the water column through flocculation, coagulation and sedimentation (Förstner and Wittman 1979, Huh et al. 1992, Hatje et al. 2003). Under suitable conditions, contaminants may accumulate in sediment to such high concentrations that they adversely affect benthic communities. In addition to these environmental concerns there are several pragmatic reasons for focusing more attention on contaminants in sediment than in the water column. The low and often highly variable concentrations of contaminants in the water column due to variations in turbulence and mixing and variable anthropogenic inputs mean that only a snapshot of water quality status and trends is gained. Important contamination events may be missed. Analysis of sediment provides a far more conservative, spatially and temporally integrated measure of contamination problems. Furthermore, contaminant concentrations in sediment are usually orders of magnitude higher than in the overlying water column (up to a million times more) and this makes detection and measurement in the laboratory easier.

Baseline metal concentrations that have been defined for sediment from KwaZulu-Natal coastal waters by the Coastal Systems research group of the CSIR are used to interpret whether metal concentrations in sediment samples represent contamination. Interpretation is supported by concurrent measurement of the grain size composition and total organic content of sediment, since these variables have a strong bearing on natural and contaminant metal concentrations in sediment. Additionally, effluent is usually rich in particulate organic matter, which may accumulate on the seabed and exert adverse effects on benthic communities. Measurement of the total organic content of sediment provides important information in this context. Other indicators of sediment quality that are monitored include chemical oxygen demand and the presence of a suite of organic chemicals, many of which are regulated under the Stockholm Convention for Persistent Organic Pollutants. The presence of faecal indicator bacteria in sediment is also used as an indicator of recent exposure to effluent derived particulate material. The toxicity of porewater is measured to determine whether toxic effects are manifesting. Porewater ammonia concentrations are measured to determine whether ammonia is contributing to the toxicity, since ammonia is highly toxic at elevated concentrations to marine organisms.

1.4.4. Benthic Macrofauna

The most important concerns in any situation where effluent is discharged to a receiving water is whether the ecology of the receiving water and the health of human users of the water is unacceptably compromised. Effluent discharge can impact the ecology of the receiving water in numerous ways. These include changes in water column primary productivity, changes in benthic invertebrate

community structure and composition, and the accumulation of contaminants in the tissue of fish and shellfish, which not only affects the health of these organisms but may also affect the health of animals and humans that consume them.

Whereas sediment chemistry provides a screening level assessment of potential adverse effects of effluent discharge, the monitoring of biological communities provides a direct measure of the effects. The Durban outfalls monitoring programme places considerable emphasis on analysing the structure and composition of benthic macrofaunal communities as an indicator of the ecological impact of the effluent discharges. Benthic macrofauna are invertebrate organisms greater than 1 mm in size that live in or on the sediment. Benthic macrofauna are widely used as indicators of environmental condition, for a variety of reasons. First, they are generally sessile or have limited mobility. This means that they are compelled to respond to local conditions *in situ* and cannot simply move away when presented with a stress. Second, they comprise a range of taxonomic groups with varying sensitivities to pollution. They thus provide a broad, integrated measure of impact at the community level. Third, most macrobenthic taxa have life spans that extend over months to years. This renders them appropriate for measuring the 'long-term' impact of a disturbance.

The structure and composition of marine benthic macrofaunal communities is influenced by many factors. These include abiotic factors, such as sediment grain size, salinity and temperature, as well as biotic factors, such as the availability of food, competition and predation. A major challenge in environmental monitoring is to distinguish between naturally occurring and anthropogenically induced changes to benthic macrofaunal communities. This is best achieved by the comparison of communities at putatively impacted sites to those at reference sites. While benthic community data has limitations, when properly analysed they remain the most ecologically relevant line of evidence regarding possible impacts on the benthic environment (McPherson et al. 2008).

The status of benthic macrofaunal communities in the vicinities of the Central Works and Southern Works outfalls and at reference sites is examined using a variety of univariate and multivariate statistical procedures. The process is aimed at revealing the nature and extent of any aberrations in community structure and composition, and by overlaying physical and chemical variables measured in sediment, gaining an understanding of the most likely factors structuring the communities. The nature of marine benthic macrofaunal communities varies between regions and there are no universally acceptable criteria for normality. Comparisons with reference stations are, therefore, of particular importance.

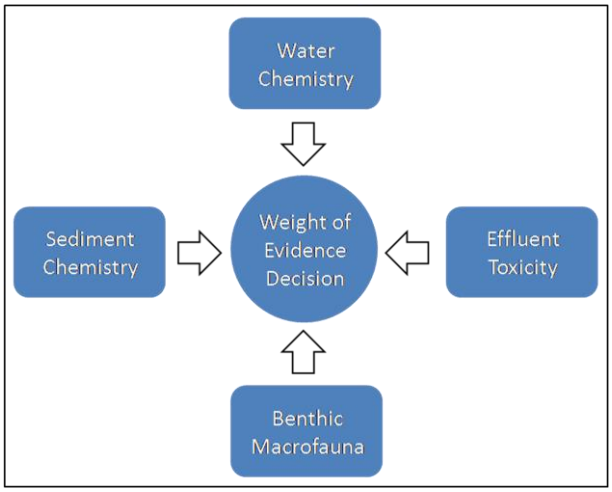

Figure 1.2. The Durban outfalls monitoring programme comprises four main components. The findings for these components are used in a weight of evidence approach to identify the impact of effluent discharge.

1.4.5. Weight of Evidence Assessment

The individual lines of evidence provided by monitoring effluent toxicity, water and sediment quality, and benthic community structure and composition are used in a weight of evidence approach to identify the impact of effluent discharge through the Central Works and Southern Works outfalls on the marine receiving water (Figure 1.2). Weighting criteria have not been defined for each line of evidence. Rather, best professional judgement is used to reach a conclusion. This said, particular emphasis is placed on benthic macrofaunal community structure and composition as an indicator of impact.

1.5. Monitoring Programme Sampling Design

The sampling design for the Durban outfalls monitoring programme has evolved considerably over the past 43 years, to accommodate changing

views, new measurement technologies, and new statistical approaches to data analysis. The continual review of the sampling design for any long-term environmental monitoring programme is essential, since this ensures that the objectives of the programme are being met in an effective, scientifically defensible and cost efficient manner within an ever-changing world. New concerns and needs are often identified by stakeholders, and wherever possible and appropriate these should be addressed through the monitoring programme. This said, the retention of key components of the sampling design is often desirable, to permit the examination of long-term trends.

The sampling design implemented in the 2009 survey of the Durban outfalls monitoring programme marked a significant change from the design used in surveys performed between 2003 and 2008. Prior to 2003 the sampling design did not include reference sites, which made it difficult to determine the type and magnitude of effluent discharge induced impacts. The rationale for and a description of the sampling design implemented in 2009 is provided in CSIR (2010). The sampling design has proved to be efficient for detecting impacts caused by effluent discharge and was consequently retained for the 2011 survey.

As discussed previously, the Durban outfalls monitoring programme is divided into several major components, namely effluent toxicity testing, water quality, sediment quality, and macrobenthic communities.

The toxicity of random samples of final effluent from the Central Works and Southern Works wastewater treatment facilities was tested monthly. The primary objective is to determine whether the minimum number of dilutions with seawater required to render the effluent non-toxic is within the theoretical minimum initial dilution for each outfall. The testing also provides an understanding of variability in the toxicity of effluent discharged through each outfall.

The main features of the sampling design for the water quality component are three stations situated to the immediate north-northeast and three stations to the immediate south-southwest of the diffuser section of each outfall, at the margin of the zone of initial dilution, and nine reference sites[2] (Figure 1.3). There were four reference sites for the Central Works outfall, one to the north-northeast and three to the south-southwest of the outfall). The closest reference sites were 2000 m to the north-northeast and south-southwest of the outfall. The reference sites were separated by a distance of 2000 m. For the Southern Works outfall there were five reference sites, three to the north-northeast and two to the south-southwest of the outfall. The nearest reference site was 800 m from the outfall, and the furthest 6000 m. As stated previously, the zone of initial dilution is a three-dimensional zone of intense mixing and dilution around the diffuser section of an outfall, within which water quality can reasonably be expected to be compromised on a regular basis (particularly in close proximity to the diffuser section). Many regulatory authorities thus stipulate that water quality at the margin of the zone of initial dilution should be compliant with water quality guidelines and other environmental quality objectives. The margin of the zone of initial dilution for each outfall was calculated as three times the average water column depth for the diffuser section. In other words, the zone of initial dilution was taken as extending about 144 m in any direction around the diffuser section of the Central Works outfall and about 177 m for the Southern Works outfall.

Discrete water samples were collected from the middle and bottom of the water column at each station and reference site. Profiles of various physical, chemical and biological variables were also measured *in situ* using a multivariable water quality monitoring sonde. The suite of variables monitored in discrete water samples and *in situ* are provided in Table 1.3. A total of 24 physical and chemical variables and biological responses (including toxicity testing) were measured, six *in situ* and the remainder in discrete water samples. A total of 967 individual measurements were made for discrete water samples and a little less than 13 000 measurements *in situ*.

[2] The term reference site is used in this report in preference to the term control site for the reason that control sites are meant to be un-impacted by anthropogenic activities. There is evidence that many of reference sites are impacted by anthropogenic activities, most notably effluent discharge. Although this technically negates use of the reference sites as points for comparison, the impact of effluent discharge on these sites as deduced from most indicators used in this monitoring programme appears to be minimal.

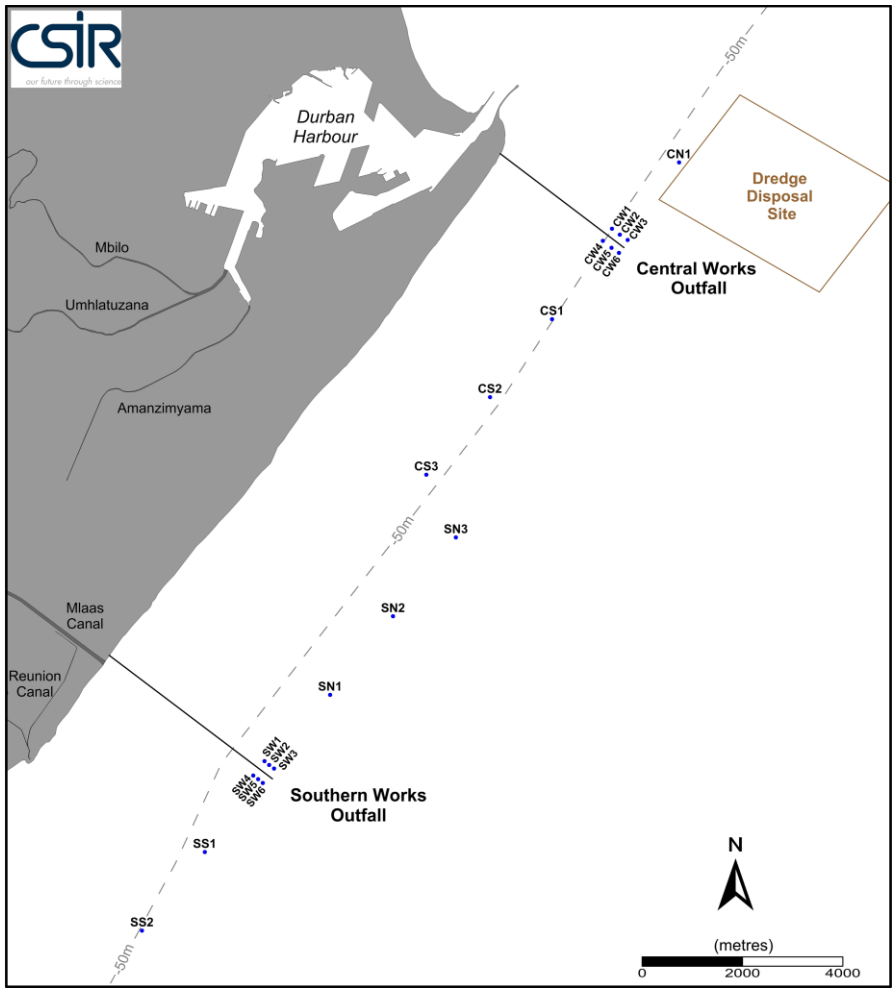

Figure 1.3. Map illustrating the sampling design for the water quality component of the 2011 survey of the Durban outfalls monitoring programme.

The primary objective of the water quality component of the monitoring programme is to determine whether the values and concentrations of physical and chemical variables at the margin of the zone of initial dilution are compliant with the South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995) and compare favourably with reference sites. Although the guidelines do not include targets for toxicity, the toxicity of discrete water samples is also tested to determine whether the effluent has been adequately diluted at the margin of the zone of initial dilution such that toxic effects do not manifest. The reference sites provide a point for comparison and over the long-term may provide data for the development of site specific baseline (or ambient) conditions for physical, chemical and biological variables and responses.

Table 1.3. Physical, chemical and biological variables measured *in situ* and analysed in discrete water samples collected for the 2011 survey of the Durban outfalls monitoring programme.

Class	Variable (*in situ*)	Class	Variable (discrete)	Class	Variable (discrete)
Conventional	Temperature	*Conventional*	Salinity	*Bacteria*	Faecal coliforms
	Salinity		pH		Faecal streptococci
	pH		Turbidity		
	Dissolved oxygen		Total suspended solids	*Toxicity*	Sea urchin fertilisation
	Turbidity		Soap, oil and grease		
	Chlorophyll-*a*	*Nutrients*	Ammonia	*Metals*	Cadmium
			Nitrite		Mercury
			Nitrate		Nickel
			Orthophosphate		Lead
			Silica		Zinc

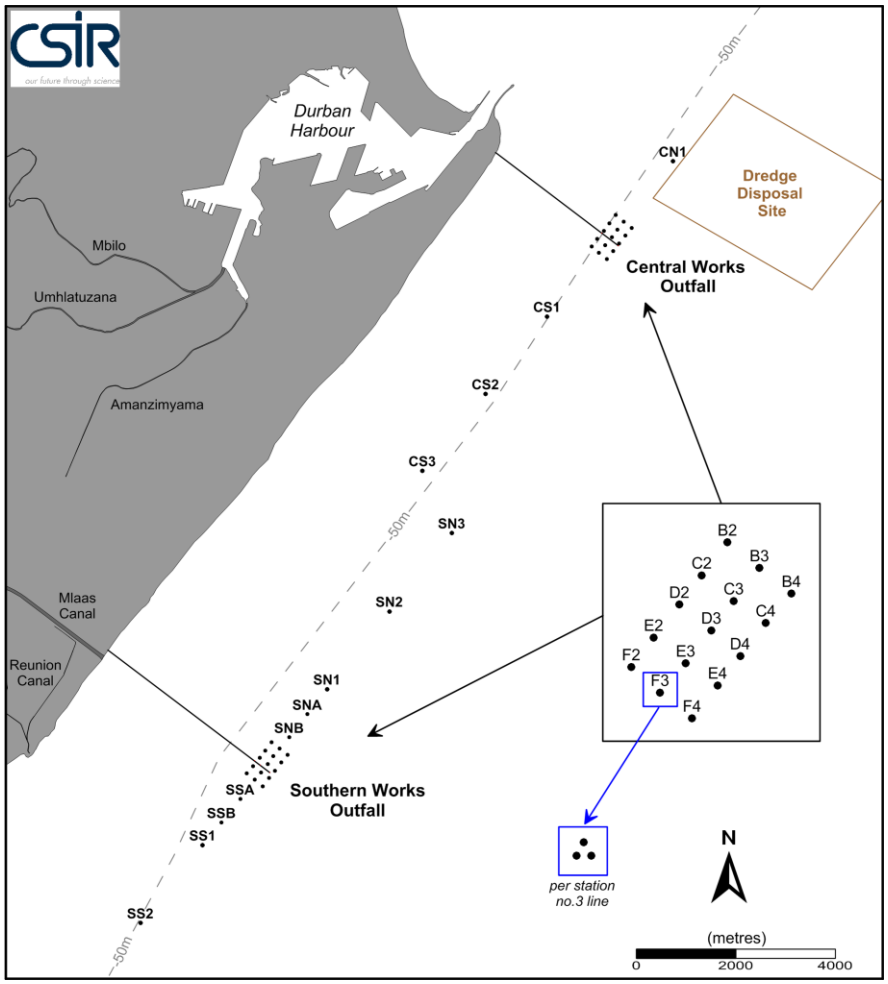

Figure 1.4. Map illustrating the sampling design for the sediment quality and benthic macrofaunal components of the 2011 survey of the Durban outfalls monitoring programme.

This will assist in determining whether values and concentrations of various variables at the margin of the zone of initial dilution fall within baseline ranges, which in many cases provide a more appropriate point for comparison than the water quality guidelines from an impact assessment perspective.

The main objective for the benthic component of the monitoring programme is to determine whether the effluent discharged is impacting sediment quality and benthic macrofaunal communities in the vicinities of the outfalls. Answers to three key questions are sought:

1. What is the spatial extent of the effluent discharge impact?
2. What is the magnitude of the effluent discharge impact?
3. Is the effluent discharge impact changing with time?

Fifteen stations that span the diffuser section of each outfall in a grid-like manner were monitored (Figure 1.4). These stations have been monitored consistently since 1994 (although the grids were of a larger spatial extent between 1994 - 2002). The grids of stations cover an area of about 0.32 km^2, with stations situated 200 m apart in the offshore and alongshore directions. The most distant stations on each grid are about 500 m from the diffuser section of the outfall.

The benthic environment at thirteen reference sites was also monitored. There were four reference sites for the Central Works outfall, one to the north-northeast and three to the south-southwest of the outfall (Figure 1.4). The closest reference sites were 2000 m to the north-northeast and south-southwest of the outfall. The reference sites were separated by a distance of 2000 m. For the Southern Works outfall there were nine reference sites, five to the north-northeast and four to the south-southwest of the outfall. The nearest reference site was 800 m from the outfall, and the furthest 6000 m. There were

more reference sites for the Southern Works outfall since adverse impacts have for some time been evident across the entire grid of stations spanning the diffuser. This suggests that impacts extend beyond the grid. The additional reference sites provide an understanding of whether the impacts are extending to 800 m or 1400 m of the outfall.

The distance between the most northerly and southerly of the reference sites was about 19 km, and this defines the Durban outfalls monitoring programme study area.

The reference sites are aligned perpendicular to the outfalls, along two 'lines'. This orientation takes into account the predominant south-southwest and north-northeast current direction off the Bluff (i.e. essentially parallel to the shoreline). Numerical modelling has demonstrated that effluent is dispersed predominantly in these directions and that the magnitude of impacts should decrease with distance from the outfalls in the same directions. The positioning of the reference sites for each outfall at different depths takes into account that the centre point of the diffuser of each outfall lies in water of a different depth (Figure 1.4, Table 1.1).

Table 1.4. Physical, chemical and biological variables and responses analysed in sediment samples collected for the 2011 survey of the Durban outfalls monitoring programme.

Class	Variable	Class	Variable
Conventional	Sediment grain size	*Organochlorine pesticides*	Lindane
			alpha-Lindane
Organic indicators	Total organic content		beta-Lindane
	Chemical oxygen demand		gamma-Lindane
	Porewater ammonia		delta-Lindane
			Heptachlor
Bacteria	Faecal coliforms		Heptachlor epoxide (cis)
	Faecal streptococci		Heptachlor epoxide (trans)
			Hexachlorobenzene
Porewater Toxicity	Sea urchin fertilisation test		Hexachlorobutodiene
			alpha-Chlordane
Metals	Aluminium		Gamma-Chlordane
	Iron		alpha-Endosulfan
	Arsenic		beta-Endosulfan
	Cadmium		alpha-Endosulfan sulfate
	Cobalt		Dieldrin
	Copper		Endrin
	Chromium		Endrin aldehyde
	Mercury		Endrin Ketone
	Nickel		Methoxychlor
	Lead		Isodrin
	Vanadium		Telodrin
	Zinc		Aldrin
			o',p' and p',p'-DDT
Polycyclic aromatic hydrocarbons	Naphthalene		o',p' and p',p'-DDD
	Naphthalene, 2-methyl-		o',p' and p',p'-DDE
	Acenaphthylene		
	Acenaphthene	*Polychlorinated biphenyls*	PCB28
	Fluorene		PCB52
	Phenanthrene		PCB101
	Anthracene		PCB118
	Fluoranthene		PCB138/163
	Pyrene		PCB153
	Benz[a]anthracene		PCB180
	Chrysene		
	Benzo[b+k]fluoranthene		
	Benzo[a]pyrene		
	Indeno[1,2,3-cd]pyrene		
	Dibenzo[a,h]anthracene		
	Benzo[ghi]perylene		

The lines of reference sites thus account for possible depth related differences in current velocity that may influence effluent dispersion and sediment grain size composition. Depth is also known to influence the composition of benthic invertebrate communities and, if not accounted for, introduces a potential confounding factor during data interpretation.

Replicate sediment samples were collected at each reference site and at each station along the central line of the grids of stations (the so-called 3 line in Figure 1.4). The replication permits a statistical comparison of data between sites as well as an estimation of small-scale variability. A single sediment sample was collected at the remaining stations (the 2 and 4 lines) of the grids. A total of 69 physical and chemical variables and biological responses were measured in sediment samples (Table 1.4), with a little under 3 950 individual measurements made (excluding benthic organisms). Every variable or response was not measured in each sediment sample. Toxicity testing and organic chemical analysis, for example, was only performed or measured in one of the replicate sediment samples collected at each reference site and from the 3 line of the grids of stations.

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Chapter 2:
Effluent Toxicity

2.1. Introduction

Although the potential ecological risk posed by effluent discharge to a receiving water can be estimated by comparing the concentrations of chemicals in the effluent and receiving water to water quality guidelines that are protective of aquatic life, this approach has several limitations. First, water quality guidelines are specific to individual chemicals, while effluent comprises a complex mixture of chemicals. Water quality guidelines thus fail to consider possible antagonistic, additive and synergistic toxic effects of effluent constituents. Second, water quality guidelines can only be applied to the chemicals for which there are guidelines. There are about 239 000 substances on the Chemical Abstracts Service list of regulated chemicals, of which an estimated 10 000 are in 'regular' use. Clearly, analysing for all of these chemicals is impossible. In any case, only a small proportion (<1%) of the chemicals have associated toxicology information.

It is in this context that effluent toxicity testing assumes an important role. Whole effluent toxicity testing measures the toxicity of effluent regardless of the chemicals present and takes into account antagonistic, additive and synergistic toxic effects. Toxicity testing puts chemical data into perspective, by acting as a measure of bioavailability. This is an important consideration since the measurement of a chemical in an environmental fate compartment does not necessarily mean that it is in a bioavailable form (i.e. can cross biological membranes). Only when it is in a bioavailable form can a chemical exert a biological effect (adverse or beneficial). Metals bound in the crystal lattice of sediment grains, for example, are in a non-bioavailable form, while those dissolved in the water column are in a bioavailable form.

To further complicate matters, because a chemical is in a bioavailable form does not mean that it will exert an effect. This can only occur if there is exposure, that is, a receptor organism actually comes into contact with the chemical. Toxicity testing takes all of these complexities into account.

There are two types of toxicity tests, namely acute and chronic. Acute toxicity tests measure the adverse effect of a toxicant to an organism over a short-term exposure relative to the organism's lifespan, and typically range in duration from about 10 minutes to 96 hrs. The most common endpoints (response) measured in acute toxicity tests are fertilisation success and mortality, with the results usually reported as percentage mortality at a given concentration or as an Effective Concentration$_{50}$ or Lethal Concentration$_{50}$ (EC_{50} and LC_{50} respectively; the concentration of effluent producing a 50% reduction in fertilisation success or mortality). These measures of toxicity are statistically determined from concentration-response plots.

Chronic toxicity tests are performed over longer periods, usually ranging from 96 hrs to about 10 days in duration, but up to 28 days for some tests. Chronic toxicity tests are designed to identify the concentration of a toxicant that is usually not lethal but that may have insidious effects, like interfering with growth rates and the development or attainment of reproduction potential (for an excellent discussion of toxicity testing see Rand and Petrocelli 1995).

This chapter analyses and discusses the findings of acute toxicity testing of final effluent from the Central Works and Southern Works wastewater treatment facilities, as determined monthly between April 2011 and March 2012. The major objective is to identify the acute toxicity of final effluent from the wastewater treatment facilities after its simulated dilution in the receiving water, to determine whether the number of dilutions required to render the effluent non-toxic is within the modelled minimum initial dilution for each outfall (see CSIR 1998). The testing also provides an understanding of the variability in toxicity of the final effluent discharged through each outfall.

2.2. Materials and Methods

2.2.1. Fieldwork

CSIR personnel collected final effluent from the Central Works and Southern Works wastewater treatment facilities on a monthly basis between April 2011 and March 2012. However, Central Works wastewater treatment facility final effluent could not be collected between May and August 2011, due to problems experienced at the facility. The effluents were returned to the laboratory and held at 4^{o}C until testing, which was always within a few days of collection.

2.2.2. Laboratory Analyses

Effluent toxicity was determined using the sea urchin fertilisation test. Toxicity is determined by comparing the fertilisation success of gametes (i.e. eggs and sperm) exposed to a test sample (i.e. a dilution of effluent) to the fertilisation success of gametes exposed to control ('clean') seawater samples. If the fertilisation success is statistically significantly different, then toxicity is evident. The sea urchin fertilisation test is used to assess the toxicity of effluent, receiving waters and sediment porewater in many regions of the world (e.g. His et al. 1999, USEPA 2002, Meric et al. 2005) and has proven to be amongst the most sensitive of tests for this purpose (e.g. Woodworth et al. 1999).

Adult sea urchins (*Echinometra mathaei* and *Tripneustes gratilla*) were collected at Vetch's Pier in Durban and maintained at ambient temperature in natural seawater in large, flow-through tanks in the laboratory. Gametes were obtained by inducing sea urchins to spawn, by injecting 1 - 2 ml of 0.5 molar potassium chloride (KCl) solution into the coelomic cavity of each test organism. Gametes from males and females were collected separately. Females were inverted over glass beakers filled with seawater and eggs were allowed to settle. Sperm was collected 'dry' in pasteur pipettes. Sea urchins that provided relatively little gametes were excluded from consideration for testing.

The quality of eggs and sperm was evaluated prior to testing, by adding diluted sperm from each male to eggs from each female in 20 ml of seawater in vials. After ten minutes, eggs were examined under a microscope for the presence of a fertilisation membrane. Combinations of eggs and sperm that did not produce at least 90% fertilisation success were excluded from consideration for testing.

Sperm was activated by exposure to seawater. One hundred microlitre aliquots of sperm suspension were then transferred to control (seawater) and effluent dilution treatments. Effluent dilutions were prepared by the addition of relevant amounts of clean seawater to effluent, with four replicates for each treatment. After ten minutes of sperm exposure, 1 ml of egg suspension was added and left for a further ten minutes. The test was then terminated by adding 100 µl of formalin. Fertilisation success was determined by microscopic examination of an aliquot of the egg suspension from each replicate. Sea urchin gamete sensitivity was assessed using the reference toxicant (positive control) formaldehyde, following a similar procedure to that outlined above.

2.2.3. Data Analysis

As stated above, toxicity testing necessitated the use of two sea urchin species, due to differences in breeding season (*T. gratilla* was used between July and October, and *E. mathaei* at other times). These species, however, exhibit different sensitivities to the copper reference toxicant (and hence also presumably to effluent of similar composition), which complicates the direct comparison of effluent toxicity across the entire testing period. In addition to inter-species differences in sensitivity, there are intra-species differences in sensitivity. Since effluent from the Central Works and Southern Works wastewater treatment facilities was tested at the same time, within month differences in effluent toxicity between the facilities was compared directly. This comparison used two indicators, namely the effective concentration (dilution) inducing a fifty percent fertilisation inhibition (EC_{50}) relative to the control treatment and the Minimum Acceptable Toxicant Dilution (MATD), which is the minimum number of effluent dilutions required to achieve a fertilisation success that is statistically not significantly different to fertilisation success in the control treatment.

It is recognised that no statistical method to determine effective concentrations causing a particular response is suitable for all data sets.

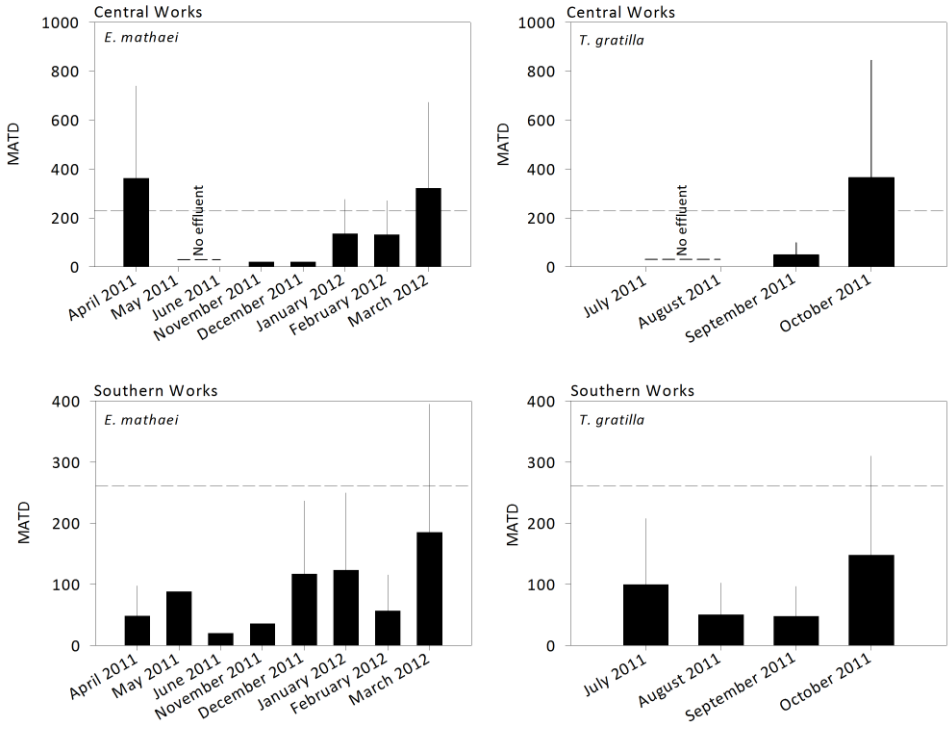

Figure 2.1. Minimum Acceptable Toxicant Dilution (MATD; +95% Confidence Interval) for final effluent from the Central Works and Southern Works wastewater treatment facilities, as determined from the fertilisation success of gametes of the sea urchins *Echinometra mathaei* and *Tripneustes gratilla* exposed to serial dilutions of effluent The absence of the upper 95% Confidence Interval for a testing date means that the interval could not be determined due to the low dilution required to achieve the Minimum Acceptable Toxicant Dilution. Horizontal dashed lines represent the lowest theoretical minimum initial dilution for each outfall predicted through numerical modelling (see Table 1.2).

Environment Canada (2002) recommends plotting the response data for treatments to produce an initial response curve against which the chosen method of statistical analyses provides logical agreement with the observed data. Following this the data was analysed using a statistical programme developed by Barnes et al. (2003), for determination of effective concentrations using logistic dose-response curves based on the formula:

$$\text{Response} = \text{Top}/(1+\exp(-\text{slope} \times (\text{logRate} - \text{LogEC50})))$$

The effluent dilution that resulted in 50% reduction in fertilisation (EC_{50}) and that which resulted in statistically no significant difference when compared to the control (i.e. the MATD) for each test was then determined.

The reader should note that the term 'significant' has both a scientific and a common meaning. The term 'statistically significant' is used by the scientific community to indicate that the average value of measurements made in one area or experimental treatment is different, with a stated level of confidence, to the average value of similar measurements made in another area or experimental treatment. For example, in this report the level of statistical inference is $\alpha = 0.05$, that is, a 95% level of confidence that the measurements are different and a 5% error level, that is, differences that are indicated but these in fact do not exist. The term 'statistically significant' does not however necessarily mean that the difference is 'biologically or ecologically significant'.

2.3. Results and Discussion

The EC_{50} and MATD for final effluent from the Central Works and Southern Works wastewater treatment facilities is presented in Figure 2.1 (see Appendix 2.6.1 for data). Final effluent from the Southern Works wastewater treatment facility evinced MATDs between <20 and 148. Thus, on none of the test dates did the number of dilutions required to render the effluent non-toxic exceed the lowest theoretical minimum initial dilution of the outfall (261). This indicates that there was little risk of toxicity beyond the zone of initial dilution for the

outfall. Of course a different opinion could be reached if reliance is placed on the upper 95% confidence interval. Use of the interval shows that the MATD could possibly have exceeded the minimum initial dilution in October 2011 and March 2012.

The variability in MATD for final effluent from the Central Works wastewater treatment facility was more pronounced, ranging from <20 to 364. On three (of 8) test dates the MATD exceeded the lowest theoretical minimum initial dilution for the outfall (229). This indicates that there was, on occasion, a risk of toxicity beyond the zone of initial dilution. If the upper 95% Confidence Interval is considered then toxic effects could possibly have manifested beyond the zone of initial dilution on two additional testing dates.

2.4. Conclusions

Toxicity testing of final effluent indicated a higher potential for impact beyond the zone of initial dilution for the Central Works outfall compared to the Southern Works outfall. This was contrary to expectation considering that the Southern Works wastewater treatment facility receives a high volume of industrial effluent, which was expected to reveal in a higher toxicity. Problems experienced at the Central Works wastewater treatment facility may be a reason for the generally higher and more variable toxicity recorded for final effluent from this facility.

However, it is extremely important to take cognisance of the fact that it is very seldom, and probably never, that the theoretical minimum initial dilution will be realised in the marine environment for the reason that the water column is always moving. Even a low current velocity is likely to substantially increase (by at least a few hundred) the number of initial dilutions achieved. Of interest in this context is the median and maximum initial dilution that may be achieved for the Central Works and Southern Works outfalls, which numerical modelling demonstrates (see Table 1.2) exceeds the number of dilutions required to render the effluent non-toxic.

2.5. References

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2.6. Appendices

Appendix 2.6.1. Effective dilutions for fifty percent inhibition of fertilisation (EC_{50}) and Minimum Acceptable Toxicant Dilution (MATD) for gametes of the sea urchins *Echinometra mathaei* and *Tripneustes gratilla* following exposure to serial dilutions of final effluent from the Central Works and Southern Works wastewater treatment facilities at various times during 2011 and 2012. Endpoints are given with lower and upper 95% Confidence Intervals (CI). - = could not be calculated, No effluent = no effluent could be collected.

Date	Species	Wastewater Treatment Facility	EC_{50}	Lower 95% CI	Upper 95% CI	MATD	Lower 95% CI	Upper 95% CI
April 2011	*E. mathaei*	Southern Works	34	32	36	48	46	50
		Central Works	232	228	235	362	346	378
May 2011	*E. mathaei*	Southern Works	67	-	-	88	-	-
		Central Works	No effluent					
June 2011	*E. mathaei*	Southern Works	-	-	-	<20	-	-
		Central Works	No effluent					
July 2011	*T. gratilla*	Southern Works	58	56	59	100	91	109
		Central Works	No effluent					
August 2011	*T. gratilla*	Southern Works	33	32	34	50	49	52
		Central Works	No effluent					
September 2011	*T. gratilla*	Southern Works	36	31	42	48	46	49
		Central Works	37	32	43	49	48	50
October 2011	*T. gratilla*	Southern Works	36	31	42	148	136	162
		Central Works	93	85	102	364	276	481
November 2011	*E. mathaei*	Southern Works	18	-	-	36	-	-
		Central Works	-	-	-	<20	-	-
December 2011	*E. mathaei*	Southern Works	71	70	72	117	114	120
		Central Works	-	-	-	20	-	-
January 2012	*E. mathaei*	Southern Works	72	71	74	123	119	126
		Central Works	85	83	87	135	129	142
February 2012	*E. mathaei*	Southern Works	30	30	31	56	54	59
		Central Works	86	83	89	131	122	140
March 2012	*E. mathaei*	Southern Works	119	114	125	185	163	210
		Central Works	183	179	186	334	319	350

Chapter 3
Water Quality

3.1. Introduction

Water quality monitoring in the vicinity of effluent outfalls serves three main purposes. First, measurements can be used to track the dispersion of effluent. Second, measurements can be used to determine whether the outfall is meeting design specifications and license conditions, which typically require that water quality outside of the zone of initial dilution should be compliant with water quality guidelines. Third, measurements can be used to assess the potential risk posed to aquatic organisms and humans in receiving waters inside and outside the zone of initial dilution, by comparing indicator values and concentrations to water quality guidelines protective of marine aquatic organisms and human health.

This chapter analyses and discusses water column physical, chemical and biological data for the 2011 survey of the Durban outfalls monitoring programme, specifically focusing on the protection of marine aquatic organisms. The major objectives are to assess the impact of effluent discharge on the quality of the receiving water, to determine whether water quality at the time of monitoring was compliant with the South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995), and to analyse temporal trends. Where guidelines were not available, or where these we considered inappropriate for the study area, results from water column sampling stations near the outfall were compared with results from reference sites[3].

3.2. Materials and Methods

3.2.1. Fieldwork

Discrete water samples were collected from the middle and bottom of the water column at three stations situated to the immediate north-northeast and south-southwest of the diffuser sections of the Central Works and Southern Works outfalls, at the margin of the zone of initial dilution (i.e. a total of 12 samples per outfall, Figure 3.1) on 26 May 2011. The margin of the zone of initial dilution was determined as three times the average water column depth for the diffuser section (i.e. about 144 m for Central Works outfall and 177 m for Southern Works outfall). Discrete surface water samples were also collected from the middle and bottom of the water column at each of the nine reference sites (Figure 3.1). The rationale for collecting water samples only in the middle and bottom of the water column is that numerical modelling predicts that there is no probability for the effluent surfacing in summer. Although monitoring was performed in autumn, the same probability was considered relevant to this period because of the relatively calm conditions prevalent at this time in KwaZulu-Natal.

Discrete water samples were collected using remotely triggered NIO bottles that were deployed concurrently. On retrieval of the bottles, aliquots of water were transferred to pre-cleaned and where appropriate autoclaved high density polyethylene or glass bottles and stored on ice in the field until transfer to the laboratory, where they were refrigerated (4^{o}C) until analysis. Aliquots of water were collected for the determination of microbiology[4], total suspended solids, nutrients, soap, oil and grease, metals, and toxicity.

Temperature, salinity, pH, dissolved oxygen, turbidity and chlorophyll-*a* concentration were profiled *in situ* at stations near the outfalls and at reference sites, by lowering a Yellow Springs Instrument 6900 multivariable water quality sonde through the water column. Data were logged at 2 second intervals, providing near-continuous measurements.

[3] Using South Africa's operational policy for wastewater disposal to the marine environment as benchmark (DWAF 2004), environmental quality targets must be complied with in the area beyond the initial dilution mixing zone. Instances in which this rule may not be strictly applied include microbiological variables that do not necessarily affect the health of aquatic organisms, but rather affect specific uses (e.g. recreation) that may be at a distant location from the point of discharge. Microbiological variables are also further subject to secondary dilution and decay while being transported away from the initial dilution mixing zone.

4 For the purposes of this study, this once-off analysis of microbiological parameters was primarily used as a tracer of effluent and not to assess potential impact on human health. The latter is addressed in another monitoring programme undertaken by the municipality

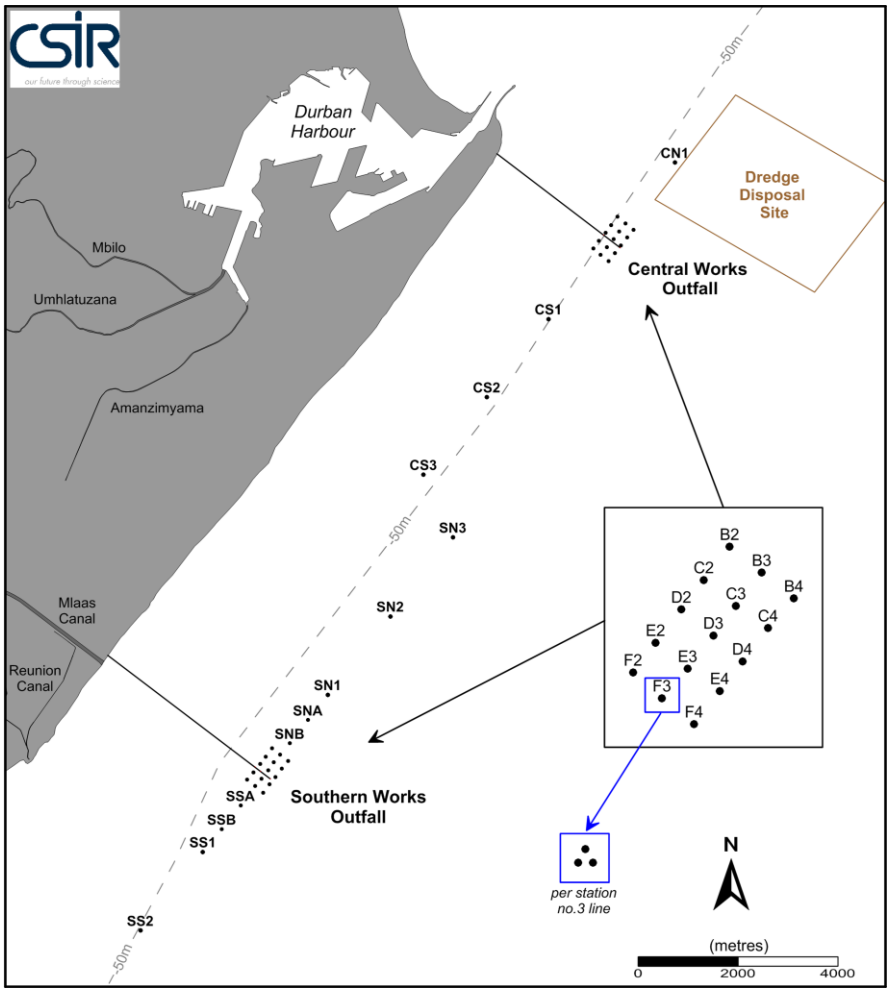

Figure 3.1. Map illustrating the positions where water quality was monitored *in situ* and water samples were collected for the 2011 survey of the Durban outfalls monitoring programme.

Probes of the sonde were calibrated a few days prior to fieldwork following the manufacturer's specifications with the exception of the chlorophyll-*a* probe, which was not calibrated. Chlorophyll-*a* concentrations were generated by default algorithms of the sondes software. The conversion of chlorophyll-*a* concentration to microalgal biomass is dependent on many factors, including the species and condition of the microalgae, and is approximate even when the sonde is calibrated. Consequently, chlorophyll-*a* concentration should be considered a relative rather than true measure of microalgal biomass. The major advantage of *in situ* instrumentation is the ability to measure near-continuous profiles of water column physical, chemical and biological variables, eliminating the need to collect discrete water samples at arbitrarily defined depths that may or may not encapsulate small yet important clines in the variables. The major disadvantage of such instrumentation, however, is the lack of (reliable) probes for measuring many variables that are usually of interest for water quality assessment (e.g. metal and nutrient concentrations) and accounts for the need to collect discrete water samples.

3.2.2. Laboratory Analyses

3.2.2.1. Accredited Laboratories

The analysis of discrete water samples was performed in environmental chemistry laboratories at CSIR campuses in Durban and Stellenbosch. These are the only laboratories in South Africa that are accredited by the South African National Accreditation System (SANAS) for the analysis of marine water, sediment and biological tissue samples.

3.2.2.2. Faecal Indicator Bacteria

Water samples were analysed for faecal coliforms and faecal streptococci by membrane filtration techniques as described in APHA (1995). This involved passing an aliquot of the sample or a volume of an appropriate dilution through a 47 mm

diameter sterile membrane filter with a 0.45 µm pore size. The membranes were then placed on selective media and incubated.

3.2.2.3. Total Suspended Solids

Total suspended solids concentrations were determined gravimetrically. Water samples were vigorously agitated and two 100 ml aliquots then vacuum filtered through pre-dried and pre-weighed 0.45 µm pore size membrane filters. Filters were then dried at 105^{o}C for 2 hrs and re-weighed. The total suspended solids concentration was determined from the difference in the dry weight of filters before and after filtration. Total suspended solids concentrations are presented on a mg.l^{-1} basis and reflect the mean of duplicate determinations.

3.2.2.4. Nutrients

On return to the laboratory, water samples were immediately vacuum filtered through 0.45 µm pore size membrane filters. The dissolved concentrations of nitrite-N (NO_2^--N), nitrate-N (NO_3^--N), total ammoniacal- or total ammonia-N (sum of NH_3^+-N and NH_4^+-N) and orthophosphate-P (PO_4^{3-}-P) in the filtrate were measured colourimetrically, using a four-channel flow injection Bran and Luebbe AutoAnalyzer II. Nutrient concentrations are reported on a mg.l^{-1} basis.

Nitrate-N was determined by quantitative reduction to nitrite-N using an activated cadmium column, the nitrite was then quantified by diazotizing with sulphonilamide and subsequently coupling to N-(1-naphthyl)-ethylenediamine dihydrochloride to form azo-dye. This process was repeated without reduction to determine the amount of nitrite in the sample, the concentration of nitrate-N was then determined by the difference in nitrite concentrations measured pre- and post-reduction.

Total ammonia-N was quantified by its reaction with sodium phenoxide and sodium hypochlorite, using sodium nitro-prusside as a catalyst in a buffered alkaline medium. Orthophosphate-P was determined by its reaction with molybdate in an acid medium to form molybdo-phosphoric acid. This was reduced to a molybdenum blue complex using ascorbic acid.

Although orthophosphate-P is considered an inorganic form of phosphorous, the reader should note that the molybdenum method used to quantify concentrations of this element is not selective for orthophosphate (Koroleff 1983, Zhang and Berberian 1997) and a fraction of organic phosphorous is probably also incorporated in the reported concentrations.

3.2.2.5. Soap, Oil and Grease

Relatively non-volatile hydrocarbons, vegetable oils, animal fats, waxes, soaps, greases, and related materials were extracted from water samples using n-hexane. Samples were acidified and then serially extracted three times with n-hexane. The extract was dried over sodium sulphate. The solvent was then distilled from the extract and the extractable material desiccated and weighed. Oil and grease concentrations are presented on a mg.l^{-1} basis.

3.2.2.6. Metals

Dissolved metal concentrations (cadmium, mercury, nickel, lead and zinc) were determined by adding a chelating agent to water samples. The metal-chelate complex was extracted using an organic solvent, the latter then removed by heating and the metal-chelate complex subsequently dissolved in dilute acid before concentrations in solution were detected and quantified by Inductively Coupled Plasma Optical Emission or Mass Spectrometry. Metal concentrations are presented on a µg.l^{-1} basis.

3.2.2.7. Toxicity Testing

The toxicity of discrete water samples was determined using the sea urchin fertilisation test. Details on the test method are provided in Chapter 2.

3.2.3. Data Analysis

Variable values and concentrations are, where possible, assessed relative to South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995; see Table 3.1).

3.3. Results and Discussion

3.3.1. Water Column Profiles

Profiles of physical, chemical and biological variables measured *in situ* are presented in Figures 3.2 to 3.7 and Figure 3.9.

The water column at several stations was strongly

thermally stratified, with the most pronounced temperature change restricted to the lower part of the water column (i.e. below about 40 m; see Figure 3.2). At station SW1, for example, the difference between the highest and lowest temperatures was 7.50°C, with most of this difference due to a sharp decrease in temperature below about 40 - 45 m. Stratification was, interestingly, not consistent across the study area. At reference site CN1, for example, the temperature was remarkably stable through the water column, varying by only 0.05°C. The difference between this situation and that described above for station SW1 is obvious.

Because water masses of a vastly different temperature have a different density they tend not to mix. Effluent discharged at depth becomes denser as it is diluted with seawater and eventually reaches neutral buoyancy. The significance is that if neutral buoyancy is reached below the thermocline then the effluent may be trapped. In this situation secondary dilution and dispersion is slower and spatially more extensive (driven by horizontal flows) compared to the situation when effluent ascends to (near) surface waters, where turbulent mixing is generally far more pronounced (driven by both horizontal and vertical flows). As discussed below, microbiological data provide evidence that the effluent was not trapped below the thermocline.

Salinity anomalies often provide a strong tracer of effluent in marine receiving waters since effluent is essentially comprised of freshwater. There were interesting anomalies for salinity in the water column along both lines of sites. These anomalies have not previously been detected in the receiving water and it is difficult to provide an adequate explanation on their occurrence and to relate the anomalies to effluent discharge. The reason is that anomalies of a similar magnitude were evident at reference sites and at two distinct depths (Figure 3.3). There were minor anomalies between about 16 - 24 m at the Central Works outfall and fairly pronounced anomalies between about 8 - 20 m for the Southern Works outfall, in addition to anomalies in the lower part of the water column. If the anomalies were due to effluent discharge then it would have been expected that they would be most intense near the outfalls, and then decrease in intensity with distance and in the direction that effluent was being dispersed. This was clearly not the case.

None of the other variables showed anomalies that

Table 3.1. South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment; DWAF 1995).

Parameter	Target Value/Concentration
Salinity	33 - 36
pH	7.3 - 8.2
Dissolved oxygen	Should not fall below 5 $mg.l^{-1}$ (99% of the time) and below 6 $mg.l^{-1}$ (95% of the time).
Turbidity	Turbidity should not reduce the depth of the euphotic zone by more than 10% of background levels measured at a comparable control site.
Suspended solids	The concentration of suspended solids should not be increased by more than 10% of the ambient concentration.
Floating matter	Water should not contain floating particulate matter, debris, oil, grease, wax, scum, foam or any similar floating materials and residues from land-based sources in concentrations that may cause nuisance.
Ammonia	600 $\mu g.l^{-1}$ as N
Nutrients	Target for the South African coastal zone: Waters should not contain concentrations of dissolved nutrients that are capable of causing excessive or nuisance growth of algae or other aquatic plants or reducing dissolved oxygen concentrations below the target range indicated for dissolved oxygen.
Cadmium	4 $\mu g.l^{-1}$
Copper	5 $\mu g.l^{-1}$
Mercury	0.3 $\mu g.l^{-1}$
Nickel	25 $\mu g.l^{-1}$
Lead	12 $\mu g.l^{-1}$
Zinc	25 $\mu g.l^{-1}$

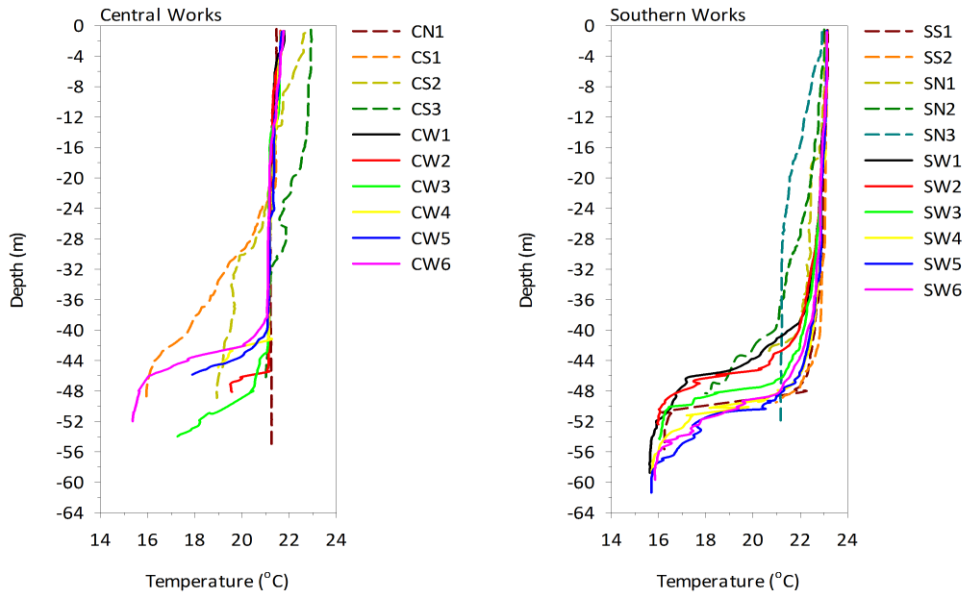

Figure 3.2. Profiles of temperature for the 2011 survey of the Durban outfalls monitoring programme.

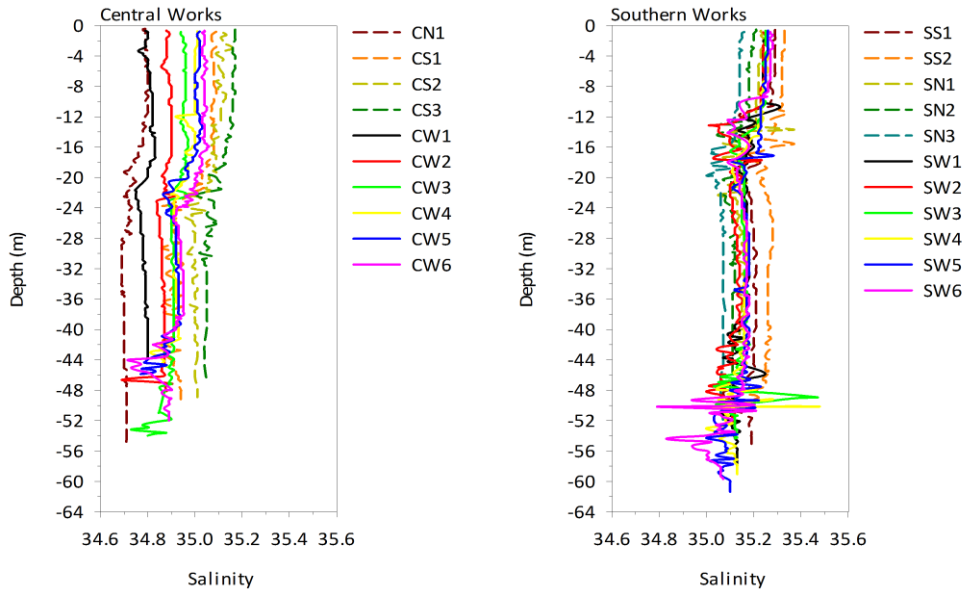

Figure 3.3. Profiles of salinity for the 2011 survey of the Durban outfalls monitoring programme.

could be attributed to effluent discharge with even a moderate level of confidence, since profiles at stations near the outfalls were similar to profiles at reference stations (Figures 3.4 to 3.7 and Figure 3.9).

The South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995) provide target values and concentrations for many of the variables measured *in situ* (see Table 3.1). The guidelines specify that the maximum acceptable non-natural variation in ambient water temperature is ±1°C. It will, however, be extremely difficult to determine whether a variation in temperature of this scale, and even a somewhat wider scale in the water column near the outfalls is attributable to effluent discharge. This is due to the temperature of the receiving water being naturally variable, both horizontally and vertically. There is consequently little point assessing water column temperature against the guideline.

Salinity at all stations was within the target range of 33 to 36. The pH through most of the water column along both lines of sites exceeded the upper limit (8.2) of the target range (Figure 3.4). Nothing should be read into this non-compliance since pH's of this

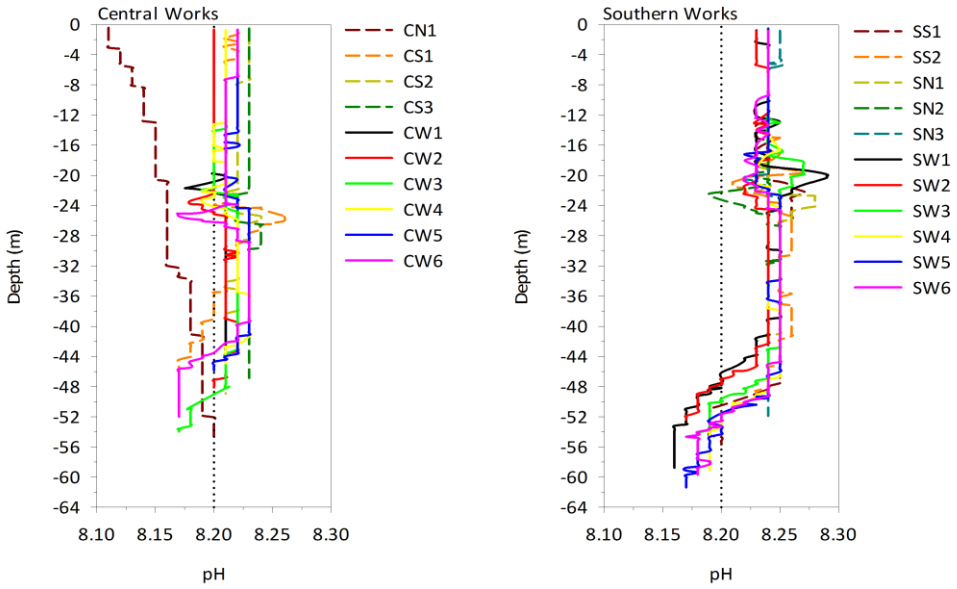

Figure 3.4. Profiles of pH for the 2011 survey of the Durban outfalls monitoring programme. The vertical stippled line represents the upper limit of the pH target range for South African coastal waters (see Table 3.1).

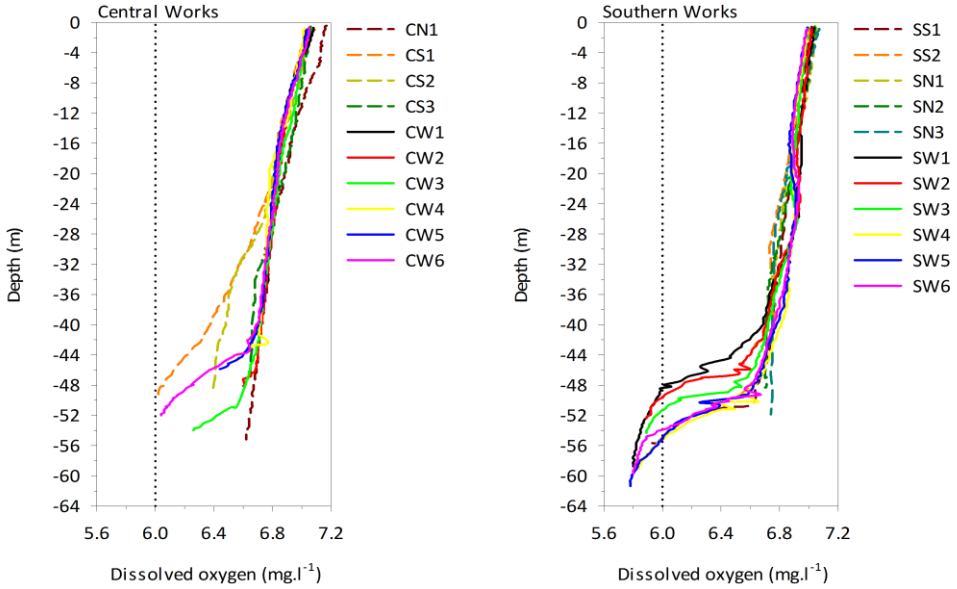

Figure 3.5. Profiles of dissolved oxygen for the 2011 survey of the Durban outfalls monitoring programme. The vertical stippled line represents the limit below which dissolved oxygen concentrations in South African coastal waters should not fall 95% of the time (see Table 3.1).

order are routinely measured in nearshore marine waters off the KwaZulu-Natal coastline, including previous surveys of the Durban outfalls monitoring programme. In other words, the upper limit of the generic target range for pH is overprotective. Some stakeholders have criticised scientists from the Coastal Systems research group of the CSIR for making this statement in previous survey reports, contending that the scientists are essentially deciding when the guidelines are appropriate. This is not the case, since this conclusion is based on a comprehensive understanding of water quality in KwaZulu-Natal coastal waters. An analysis of marine water quality guidelines from many regions of the world demonstrates that the upper limit for pH is in the order of 8.5 - 8.7 (e.g. Canada: CCME 1999, United States: USEPA 1999). This takes into account that the pH of marine waters in many regions of the world is in the order of 8.1 - 8.3 but also makes allowance for increases above about 8.3 due to natural processes (e.g. high microalgal productivity).

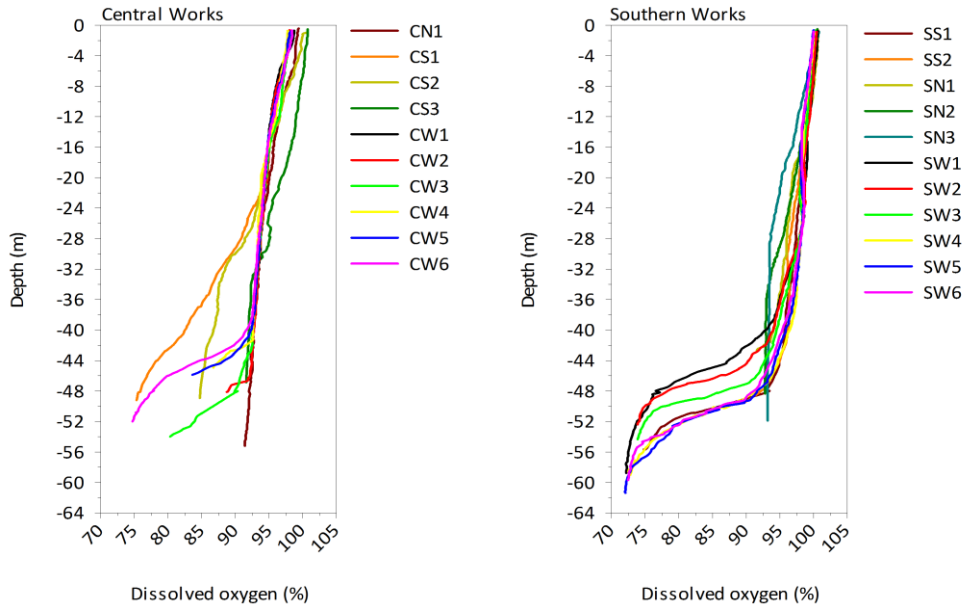

Figure 3.6. Profiles of dissolved oxygen saturation for the 2011 survey of the Durban outfalls monitoring programme.

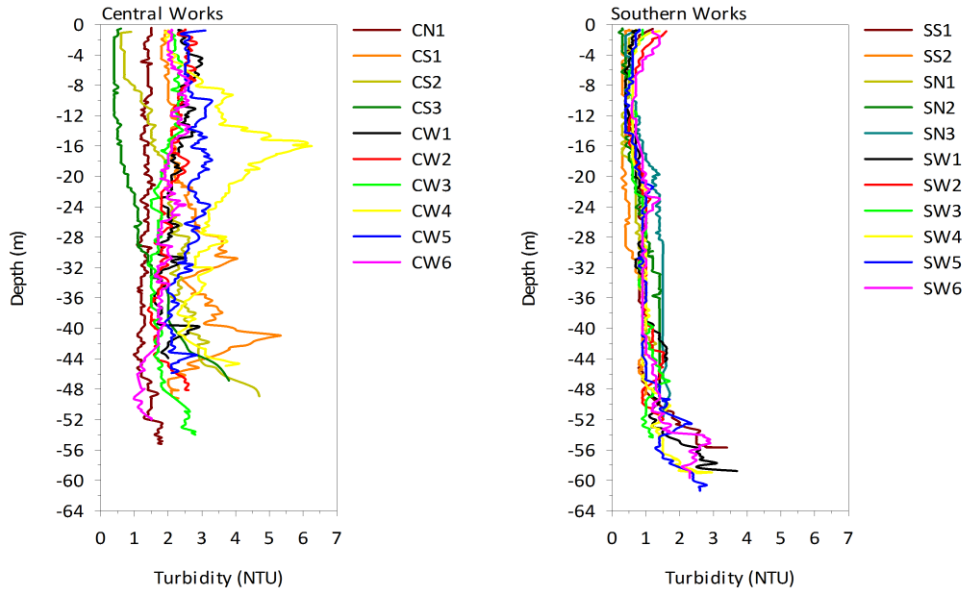

Figure 3.7. Profiles of turbidity for the 2011 survey of the Durban outfalls monitoring programme.

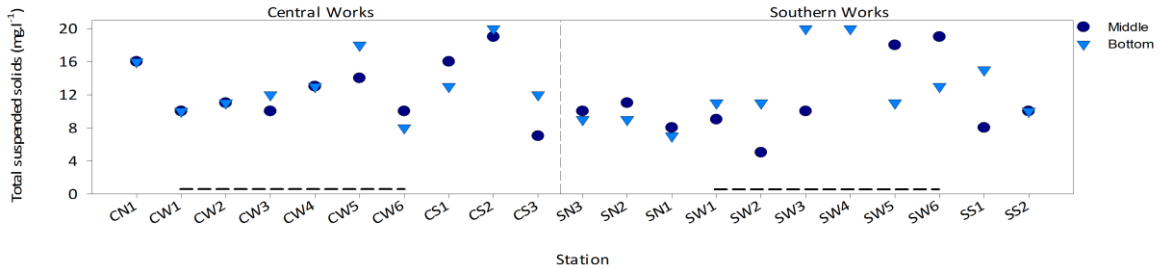

Figure 3.8. Total suspended solids concentrations in water samples collected for the 2011 survey of the Durban outfalls monitoring programme. The vertical dashed line separates data for the Central Works and Southern Works outfalls. The horizontal dashed lines denote stations situated at the margin of the zone of initial dilution for each outfall.

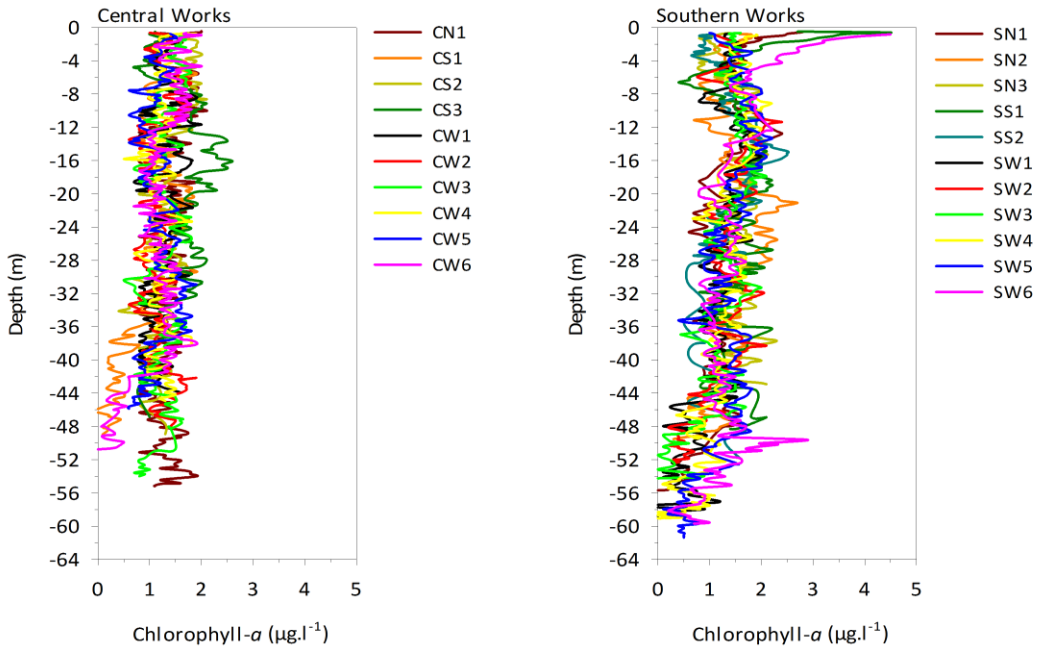

Figure 3.9. Profiles of chlorophyll-*a* in the 2011 survey of the Durban outfalls monitoring programme.

Scientists from the Coastal Systems research group are in fact of the opinion that the lower limit of the target range (7.3) is considerably underprotective and should not be lower than 8.0 for KwaZulu-Natal marine waters.

Dissolved oxygen concentration through the water column at Central Works stations and reference sites was compliant with relevant guidelines (Figure 3.5, Table 3.1). At Southern Works stations and reference sites, dissolved oxygen concentration in the lower part of the water column fell below 6 $mg.l^{-1}$, which must be met 95% of the time. The decrease in dissolved oxygen concentration below about 40 m is clearly linked to the stratification discussed previously, as revealed by the similar profiles measured at all sites, including the reference sites. Further, since a dissolved oxygen concentration of ≥5 $mg.l^{-1}$ is generally regarded as protective of aquatic life (USEPA 2003) it is unlikely that the 'low' dissolved oxygen concentrations were ecologically stressful.

The guidelines state that 'turbidity should not reduce the depth of the euphotic zone by more than 10% of background levels measured at a comparable control site' (Table 3.1). Scientists from the Coastal Systems research group of the CSIR also consider the turbidity guideline to be overly restrictive in situations where the turbidity is low (i.e. below about 10 - 20 NTU). For example, a 10% change to a background turbidity of 10 NTU amounts to 1 NTU. Whether such a small change is ecologically meaningful is debatable. Furthermore, it would be extremely difficult to attribute a change of as little as 1 NTU to an anthropogenic activity when it is considered that the turbidity of the water column near the outfall frequently varies naturally through the water column at a scale far greater than this and in an often unpredictable manner. The turbidity of the water column at stations near the outfalls was, however, low and within the range measured at reference stations (Figure 3.7).

3.3.2. Discrete Water Samples

Faecal indicator bacteria are amongst (if not) the most useful indicators for tracing effluent. The reason is that the bacteria are found only in the faeces of mammals and birds (Dufor 1977). Since there are no bird or mammal populations permanently resident at or near the Central Works and Southern Works outfalls, and the outfalls discharge effluent a substantial distance offshore and at depth, the only (or at least most significant) source of faecal indicator bacteria in the water column near the outfalls must by implication be effluent.

With one exception, faecal indicator bacteria colony forming unit counts at the Central Works stations

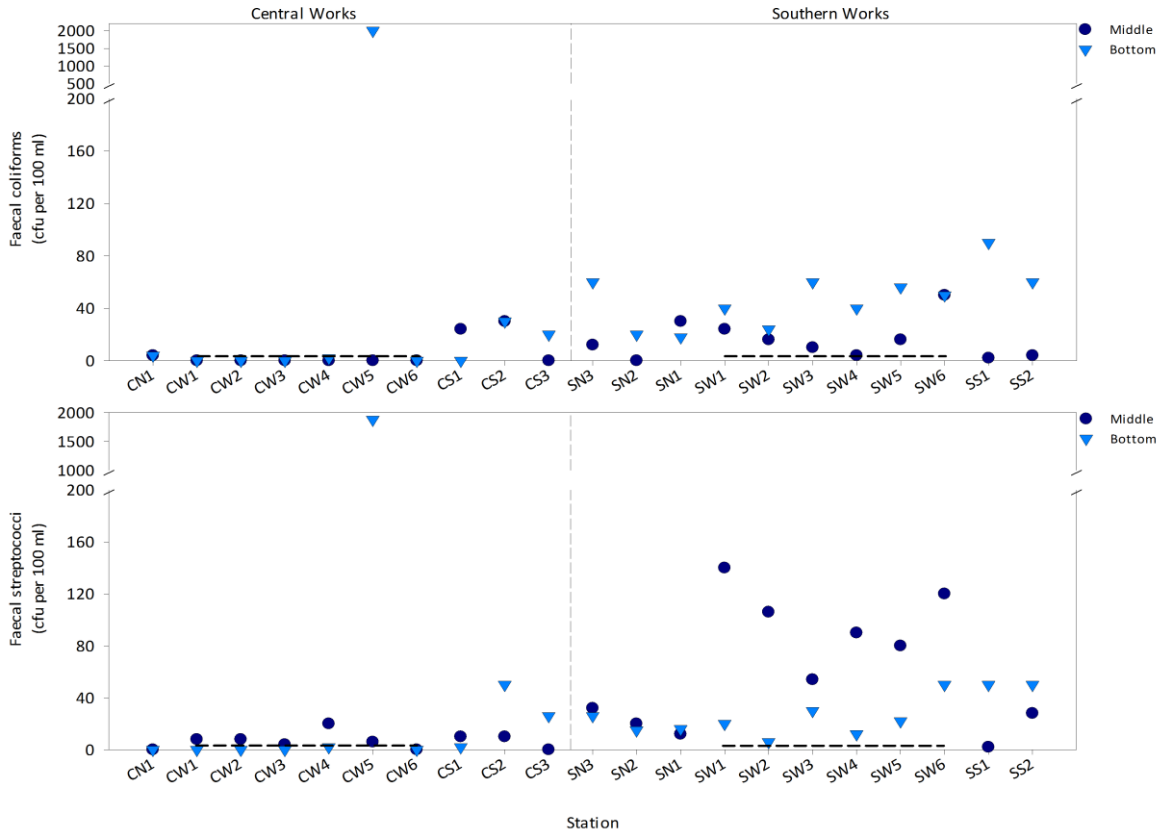

Figure 3.10. Faecal coliform and faecal streptococci bacteria colony forming unit counts in water samples collected for the 2011 survey of the Durban outfalls monitoring programme. The vertical dashed line separates data for the Central Works and Southern Works outfalls. The horizontal dashed lines denote stations situated at the margin of the zone of initial dilution for each outfall.

and at reference station CN1, situated 2 000 m to the north-northeast of the outfall, were very low (Figure 3.10). In fact, the majority of samples revealed no bacteria. The notable exception was station CW5, where very high counts in the bottom water sample were detected and is almost certainly an effluent signal. Colony forming unit counts were slightly higher at reference stations situated to the south-southwest of the outfall, but the counts were still low (Figure 3.10).

Faecal indicator bacteria colony forming unit counts at Southern Works stations and reference sites were generally higher compared to the Central Works stations and reference sites, but were nevertheless also low (Figure 3.10). Interestingly, the trend for faecal coliforms and faecal streptococci was different for stations in the immediate vicinity of the outfall. Thus, faecal coliform counts in bottom water were generally higher than in mid-water, with the trend reversed for faecal streptococci. Faecal streptococci counts in mid-water were somewhat higher compared to the reference stations and provide the clearest signal of the effluent plume (Figure 3.10).

If the previous contention that the only source of bacteria to the study area is effluent discharge, then it is clear that while effluent signals were evident across most of the study area they were more pronounced in the immediate vicinities of the outfalls.

The concentrations of nutrients, soaps, oils and grease, and metals in the discrete water samples showed no trends that could conclusively be attributed to effluent discharge (Figure 3.11 - 3.13). In other words, concentrations in the immediate vicinities of the outfalls were within ranges measured at reference sites.

With one exception, total ammonia-N and metal concentrations in discrete water samples collected at the margin of the zone of initial dilution and at reference sites were compliant with the South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995). The

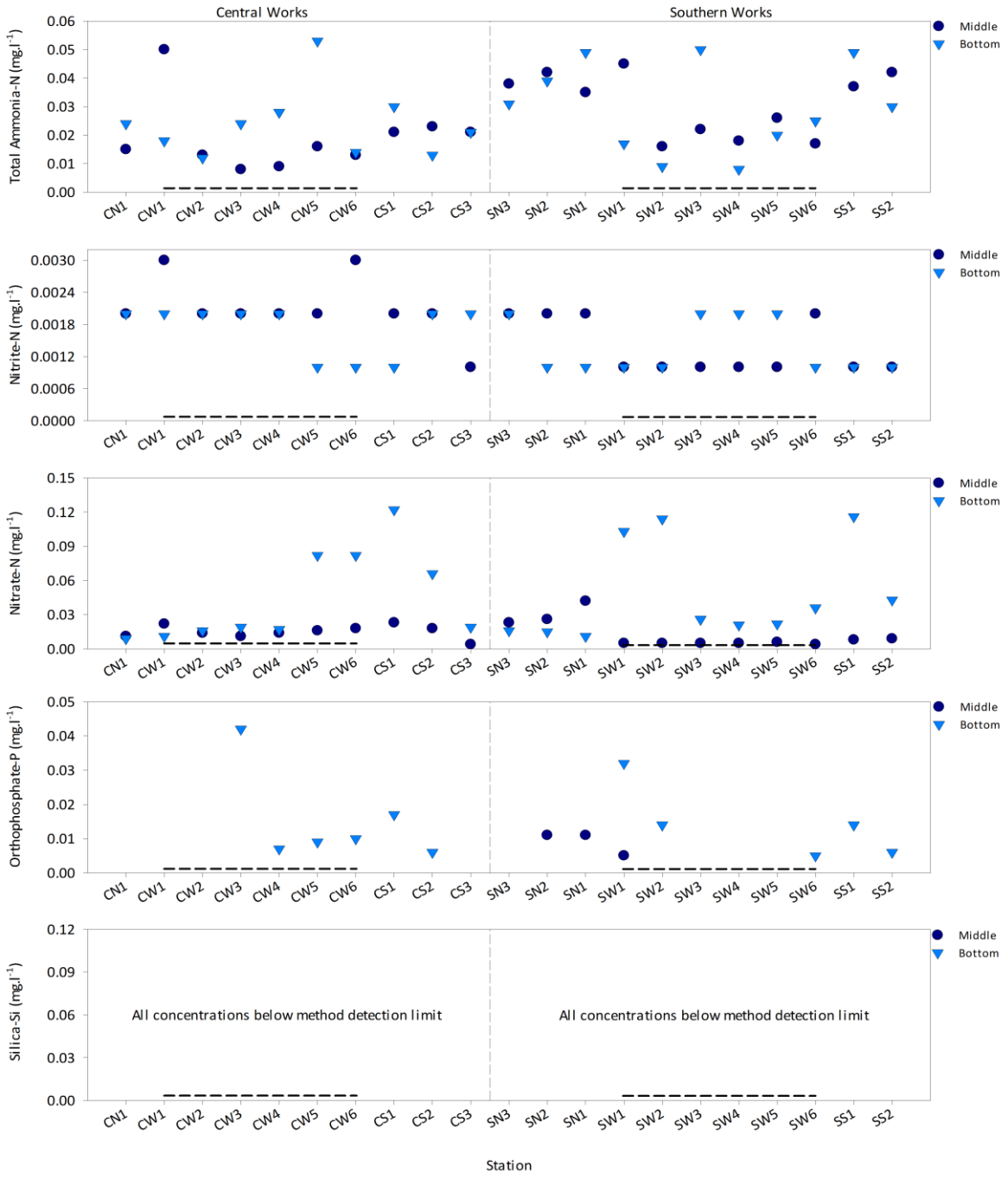

Figure 3.11. Nutrient concentrations in water samples collected for the 2011 survey of the Durban outfalls monitoring programme. The vertical dashed lines separate data for the Central Works and Southern Works outfalls. The horizontal dashed lines denote stations situated at the margin of the zone of initial dilution for each outfall. Where no data points are evident for a station the concentration was below the method detection limit.

exception was for copper, which was present at a concentration exceeding the guideline in two samples, both collected at reference sites to the south-southwest of the Central Works outfall (Figure 3.13). One of these concentrations was in fact very high. It is, however, difficult to interpret these concentrations considering that copper was present at concentrations above the method detection limit in so few samples and in a spatially haphazard manner.

None of the water samples was toxic to sea urchin gametes (Figure 3.14).

3.4. Comparison to Previous Surveys

Since the sampling design used in the 2011 survey of the Durban outfalls monitoring programme was first implemented in 2009, comparison of the values and

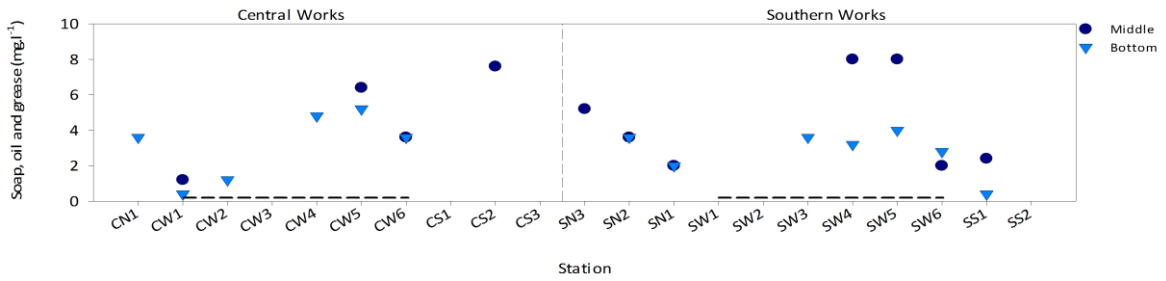

Figure 3.12. Soap, oil and grease concentrations in water samples collected for the 2011 survey of the Durban outfalls monitoring programme. The vertical dashed line separates data for the Central Works and Southern Works outfalls. The horizontal dashed lines denote stations situated at the margin of the zone of initial dilution for each outfall. Where no data points are evident for a station the concentration was below the method detection limit.

concentrations of physical, chemical and biological variables in discrete water samples is only made for this period. Comparison is made between outfall and reference stations regardless of the collection depth. The purpose of the comparison is to provide perspective on trends between surveys.

Comparison is made with the aid of cumulative frequency plots (Figure 3.15). The plots present ranked distributions in ascending order of the values and concentrations. The lowest value and concentration measured is displayed on the extreme left of the plot and the highest concentration on the extreme right. The position occupied within the distribution allows for comparison between years, while the shape of the distributions allows for comparison between outfall stations and reference sites.

Salinity and pH trends were comparable between surveys and outfalls (Figure 3.15). Trends for nutrients were also comparable, although orthophosphate-P concentrations at two stations in 2011 were the highest and second highest measured since 2009 (Figure 3.15). Both of these concentrations were measured at outfall stations, one at the Central Works outfall and the other at the Southern Works outfall. Faecal indicator bacteria colony forming unit counts in 2011 were generally amongst the lowest since 2009. Only two metals were detected at concentrations exceeding the method detection limit frequently enough to make a sensible comparison, namely copper and zinc. Two copper concentrations in 2011 were the highest and second highest measured since 2009 (Figure 3.15). As stated previously, both concentrations were measured at reference sites. There was little difference in the distribution of zinc concentrations between surveys with the exception of two high zinc concentrations in the 2010 survey.

3.5. Conclusions

Various physical, chemical and biological variables and responses were measured *in-situ* and in discrete water samples collected on a single occasion in May 2011. The data do not, therefore, provide a synoptic understanding of the variability of the variables and responses in the study area but only at the time of monitoring.

Faecal indicator bacteria counts provided an effluent signal. None of the other indicators provided signals that could confidently be attributed to effluent discharge. None of the water samples was toxic to sea urchin gametes. The values and concentrations of the majority of physical and chemical variables were compliant with the South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995) for which targets are available. There were non-compliances for pH, dissolved oxygen concentration and copper. For pH the non-compliances were spurious because the upper limit for this guideline is too low. For dissolved oxygen, the non-compliance was due to a natural cause (i.e. water column stratification) rather than effluent discharge. Two copper concentrations were non-compliant with the relevant guideline. One of these concentrations was in fact very high. However, it is difficult to interpret these concentrations since they were detected in samples collected a substantial distance from the outfalls, and concentrations at most stations were below the method detection limit. The trends for most variables were qualitatively similar to those for the 2009 and 2010 surveys.

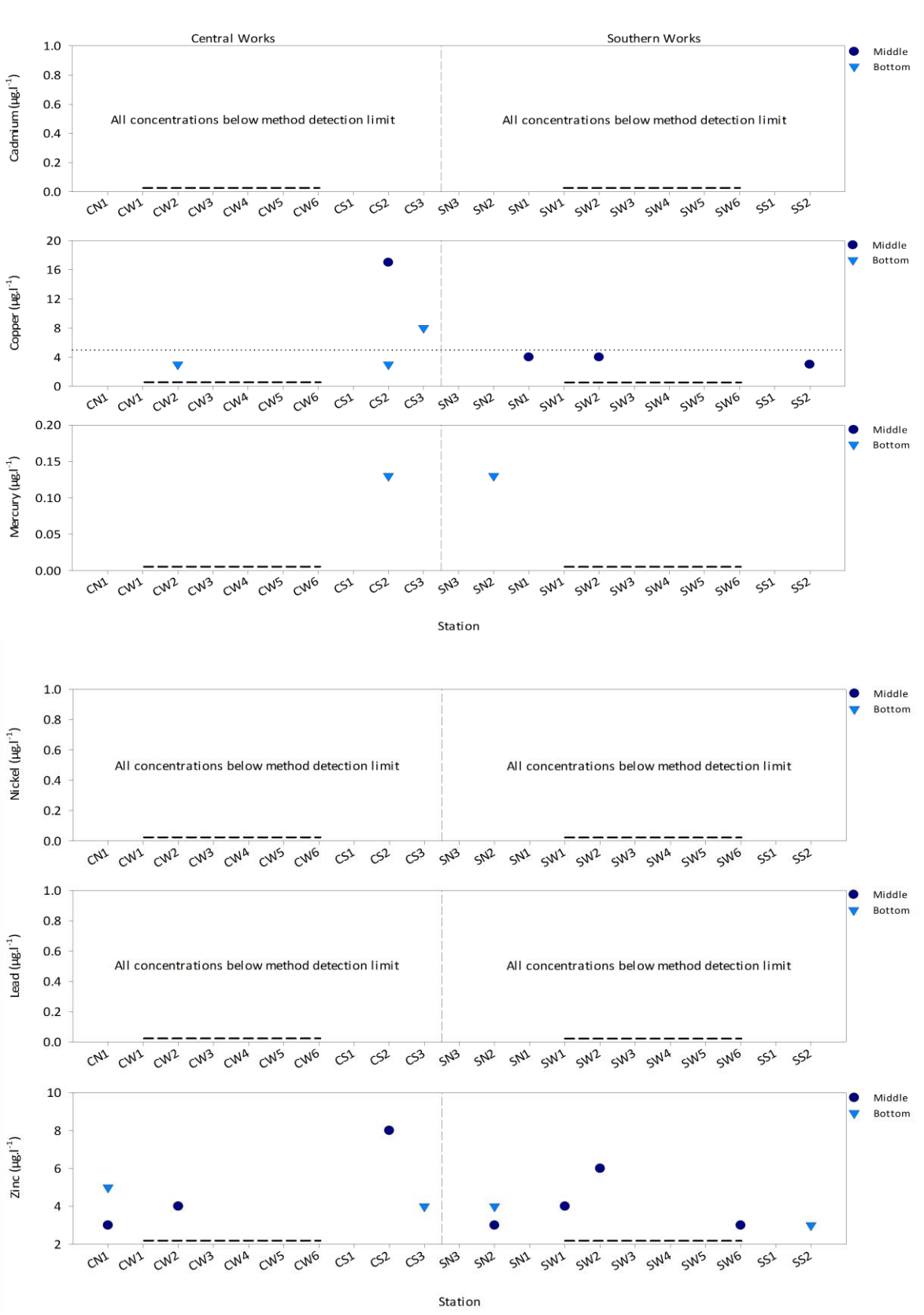

Figure 3.13. Metal concentrations in water samples collected for the 2011 survey of the Durban outfalls monitoring programme. The vertical dashed lines separate data for the Central Works and Southern Works outfalls. The horizontal dashed lines denote stations situated at the margin of the zone of initial dilution for each outfall. Where no data points are evident for a station the concentration was below the method detection limit. The horizontal stippled line in the cadmium plot represents the target for South African coastal waters (see Table 3.1).

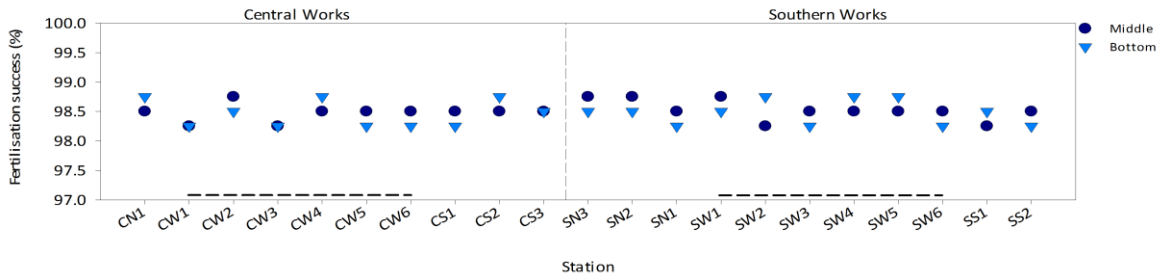

Figure 3.14. Average fertilisation success of sea urchin gametes exposed to water samples collected for the 2011 survey of the Durban outfalls monitoring programme (note that since standard deviations overlap they are not included for the sake of clarity). The vertical dashed line separates data for the Central Works and Southern Works outfalls. The horizontal dashed lines denote stations situated at the margin of the zone of initial dilution for each outfall.

3.6. References

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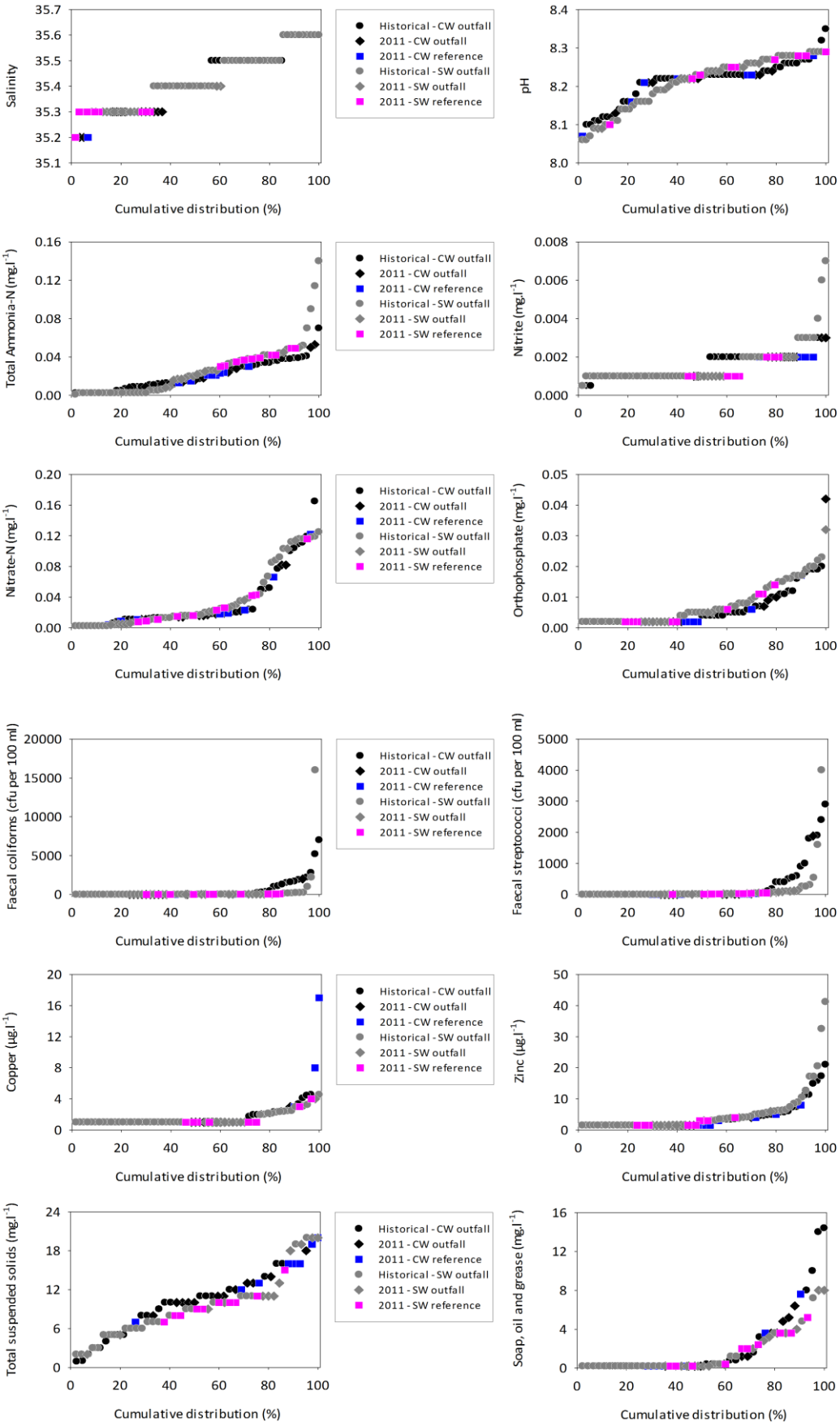

Figure 3.15. Comparison of the cumulative distribution for various physical, chemical and biological parameters measured in discrete water samples collected for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme.

3.7. Appendices

Appendix 3.7.1. Faecal indicator bacteria counts, total suspended solids concentrations, soap, oil and grease concentrations, and results of toxicity testing of discrete water samples collected for the 2011 survey of the Durban outfalls monitoring programme. < = concentration below indicated method detection limit.

Station	Depth	Faecal coliforms cfu/100 ml	Faecal streptococci cfu/100 ml	Total Suspended Solids ($mg.l^{-1}$)	Soap, Oil and Grease ($mg.l^{-1}$)	Sea Urchin Average Fertilisation Success (%)[5]	Sea Urchin Fertilisation Standard deviation
CN1	Middle	4	0	16.0	<0.40	98.50	0.58
CN1	Bottom	4	0	16.0	3.60	98.75	0.50
CW1	Middle	0	8	10.0	1.20	98.25	0.50
CW1	Bottom	0	0	10.0	0.40	98.25	0.50
CW2	Middle	0	8	11.0	<0.40	98.75	0.50
CW2	Bottom	0	0	11.0	1.20	98.50	0.58
CW3	Middle	0	4	10.0	<0.40	98.25	0.50
CW3	Bottom	0	0	12.0	<0.40	98.25	0.50
CW4	Middle	0	20	13.0	<0.40	98.50	0.58
CW4	Bottom	2	2	13.0	4.80	98.75	0.50
CW5	Middle	0	6	14.0	6.40	98.50	0.58
CW5	Bottom	2000	1880	18.0	5.20	98.25	0.50
CW6	Middle	0	0	10.0	3.60	98.50	0.58
CW6	Bottom	0	0	8.00	3.60	98.25	0.50
CS1	Middle	24	10	16.0	<0.40	98.50	0.58
CS1	Bottom	0	2	13.0	<0.40	98.25	0.50
CS2	Middle	30	10	19.0	7.60	98.50	0.58
CS2	Bottom	30	50	20.0	<0.40	98.75	0.50
CS3	Middle	0	0	7.00	<0.40	98.50	0.58
CS3	Bottom	20	26	12.0	<0.40	98.50	0.58
SN3	Middle	12	32	10.0	5.20	98.75	0.50
SN3	Bottom	60	26	9.00	<0.40	98.50	0.58
SN2	Middle	0	20	11.0	3.60	98.75	0.50
SN2	Bottom	20	15	9.00	3.60	98.50	0.58
SN1	Middle	30	12	8.00	2.00	98.50	0.58
SN1	Bottom	18	16	7.00	2.00	98.25	0.50
SW1	Middle	24	140	9.00	<0.40	98.75	0.50
SW1	Bottom	40	20	11.0	<0.40	98.50	0.58
SW2	Middle	16	106	5.00	<0.40	98.25	0.50
SW2	Bottom	24	6	11.0	<0.40	98.75	0.50
SW3	Middle	10	54	10.0	<0.40	98.50	0.58
SW3	Bottom	60	30	20.0	3.60	98.25	0.50
SW4	Middle	4	90	2.00	8.00	98.50	0.58
SW4	Bottom	40	12	20.0	3.20	98.75	0.50
SW5	Middle	16	80	18.0	8.00	98.50	0.58
SW5	Bottom	56	22	11.0	4.00	98.75	0.50
SW6	Middle	50	120	19.0	2.00	98.50	0.58
SW6	Bottom	50	50	13.0	2.80	98.25	0.50
SS1	Middle	2	2	8.00	2.40	98.25	0.50
SS1	Bottom	90	50	15.0	0.40	98.50	0.58
SS2	Middle	4	28	10.0	<0.40	98.50	0.58
SS2	Bottom	60	50	10.0	<0.40	98.25	0.50

[5] Control fertilisation success = 98.50 ± 0.58%.

Appendix 3.7.1 continued. Nutrient concentrations in discrete water samples collected for the 2011 survey of the Durban outfalls monitoring programme. < = concentration below indicated method detection limit.

Station	Depth	Total Ammonia-N ($mg.l^{-1}$)	Nitrite-N ($mg.l^{-1}$)	Nitrate-N ($mg.l^{-1}$)	Orthophosphate-P ($mg.l^{-1}$)	Silica-Si ($mg.l^{-1}$)
CN1	Middle	0.015	0.002	0.011	<0.004	<0.100
CN1	Bottom	0.024	0.002	0.009	<0.004	<0.100
CW1	Middle	0.050	0.003	0.022	<0.004	<0.100
CW1	Bottom	0.018	0.002	0.011	<0.004	<0.100
CW2	Middle	0.013	0.002	0.014	<0.004	<0.100
CW2	Bottom	0.012	0.002	0.016	<0.004	<0.100
CW3	Middle	0.008	0.002	0.011	<0.004	<0.100
CW3	Bottom	0.024	0.002	0.019	0.042	<0.100
CW4	Middle	0.009	0.002	0.014	<0.004	<0.100
CW4	Bottom	0.028	0.002	0.017	0.007	<0.100
CW5	Middle	0.016	0.002	0.016	<0.004	<0.100
CW5	Bottom	0.053	0.001	0.082	0.009	<0.100
CW6	Middle	0.013	0.003	0.018	<0.004	<0.100
CW6	Bottom	0.014	0.001	0.082	0.010	<0.100
CS1	Middle	0.021	0.002	0.023	<0.004	<0.100
CS1	Bottom	0.030	0.001	0.122	0.017	<0.100
CS2	Middle	0.023	0.002	0.018	<0.004	<0.100
CS2	Bottom	0.013	0.002	0.066	0.006	<0.100
CS3	Middle	0.021	0.001	0.004	<0.004	<0.100
CS3	Bottom	0.021	0.002	0.019	<0.004	<0.100
SN3	Middle	0.038	0.002	0.023	<0.004	<0.100
SN3	Bottom	0.031	0.002	0.016	<0.004	<0.100
SN2	Middle	0.042	0.002	0.026	0.011	<0.100
SN2	Bottom	0.039	0.001	0.015	<0.004	<0.100
SN1	Middle	0.035	0.002	0.042	0.011	<0.100
SN1	Bottom	0.049	0.001	0.011	<0.004	<0.100
SW1	Middle	0.045	0.001	0.005	0.005	<0.100
SW1	Bottom	0.017	0.001	0.103	0.032	<0.100
SW2	Middle	0.016	0.001	0.005	<0.004	<0.100
SW2	Bottom	0.009	0.001	0.114	0.014	<0.100
SW3	Middle	0.022	0.001	0.005	<0.004	<0.100
SW3	Bottom	0.050	0.002	0.026	<0.004	<0.100
SW4	Middle	0.018	0.001	0.005	<0.004	<0.100
SW4	Bottom	0.008	0.002	0.021	<0.004	<0.100
SW5	Middle	0.026	0.001	0.006	<0.004	<0.100
SW5	Bottom	0.020	0.002	0.022	<0.004	<0.100
SW6	Middle	0.017	0.002	0.004	<0.004	<0.100
SW6	Bottom	0.025	0.001	0.036	0.005	<0.100
SS1	Middle	0.037	0.001	0.008	<0.004	<0.100
SS1	Bottom	0.049	0.001	0.116	0.014	<0.100
SS2	Middle	0.042	0.001	0.009	<0.004	<0.100
SS2	Bottom	0.030	0.001	0.043	0.006	<0.100

Appendix 3.7.1 continued. Metal concentrations in discrete water samples collected for the 2011 survey of the Durban outfalls monitoring programme. < = concentration below indicated method detection limit.

Station	Depth	Cadmium ($\mu g.l^{-1}$)	Copper ($\mu g.l^{-1}$)	Mercury ($\mu g.l^{-1}$)	Nickel ($\mu g.l^{-1}$)	Lead ($\mu g.l^{-1}$)	Zinc ($\mu g.l^{-1}$)
CN1	Middle	<1	<2	<0.08	<2	<2	3
CN1	Bottom	<1	<2	<0.08	<2	<2	5
CW1	Middle	<1	<2	<0.08	<2	<2	<3
CW1	Bottom	<1	<2	<0.08	<2	<2	<3
CW2	Middle	<1	<2	<0.08	<2	<2	4
CW2	Bottom	<1	3	<0.08	<2	<2	<3
CW3	Middle	<1	<2	<0.08	<2	<2	<3
CW3	Bottom	<1	<2	<0.08	<2	<2	<3
CW4	Middle	<1	<2	<0.08	<2	<2	<3
CW4	Bottom	<1	<2	<0.08	<2	<2	<3
CW5	Middle	<1	<2	<0.08	<2	<2	<3
CW5	Bottom	<1	<2	<0.08	<2	<2	<3
CW6	Middle	<1	<2	<0.08	<2	<2	<3
CW6	Bottom	<1	<2	<0.08	<2	<2	<3
CS1	Middle	<1	<2	<0.08	<2	<2	<3
CS1	Bottom	<1	<2	<0.08	<2	<2	<3
CS2	Middle	<1	17	<0.08	<2	<2	8
CS2	Bottom	<1	3	0.13	<2	<2	<3
CS3	Middle	<1	<2	<0.08	<2	<2	<3
CS3	Bottom	<1	8	<0.08	<2	<2	4
SN3	Middle	<1	<2	<0.08	<2	<2	<3
SN3	Bottom	<1	<2	<0.08	<2	<2	<3
SN2	Middle	<1	<2	<0.08	<2	<2	3
SN2	Bottom	<1	<2	0.13	<2	<2	4
SN1	Middle	<1	4	<0.08	<2	<2	<3
SN1	Bottom	<1	<2	<0.08	<2	<2	<3
SW1	Middle	<1	<2	<0.08	<2	<2	4
SW1	Bottom	<1	<2	<0.08	<2	<2	<3
SW2	Middle	<1	4	<0.08	<2	<2	6
SW2	Bottom	<1	<2	<0.08	<2	<2	<3
SW3	Middle	<1	<2	<0.08	<2	<2	<3
SW3	Bottom	<1	<2	<0.08	<2	<2	<3
SW4	Middle	<1	<2	<0.08	<2	<2	<3
SW4	Bottom	<1	<2	<0.08	<2	<2	<3
SW5	Middle	<1	<2	<0.08	<2	<2	<3
SW5	Bottom	<1	<2	<0.08	<2	<2	<3
SW6	Middle	<1	<2	<0.08	<2	<2	3
SW6	Bottom	<1	<2	<0.08	<2	<2	<3
SS1	Middle	<1	<2	<0.08	<2	<2	<3
SS1	Bottom	<1	<2	<0.08	<2	<2	<3
SS2	Middle	<1	3	<0.08	<2	<2	<3
SS2	Bottom	<1	<2	<0.08	<2	<2	3

Chapter 4
Sediment Quality

4.1. Introduction

Particulate materials that lie below the water in aquatic ecosystems, including shell hash, gravel, sand, mud and organic detritus are referred to as sediment. Sediment is an important component of aquatic ecosystems since it provides essential habitat for sediment-dwelling organisms. Sediment is also an important sink for many contaminants that are anthropogenically introduced into surface waters. Many contaminants have low water solubility and are particle reactive and once introduced into marine waters adsorb onto suspended sediment and organic matter and are in this manner 'scavenged' from the water column through flocculation, coagulation and sedimentation (De Groot et al. 1976, Förstner and Wittman 1979, Olsen et al. 1982, Huh et al. 1992, Honeyman and Santschi 1988, Mwanuzi and De Smedt 1999, Hatje et al. 2003, Moon et al. 2008). Consequently, the concentrations of most contaminants in sediment and at the sediment-water interface usually exceed those in the overlying water column by several orders of magnitude (Horowitz 1991, Daskalakis and O'Connor 1995). In hydro-dynamically low energy environments, where there is little sediment redistribution, contaminants can accumulate in sediment to concentrations that may adversely impact benthic organisms through direct toxicity or indirectly by altering community and food web structure (Chapman 1989).

Not surprisingly, the monitoring of contaminant concentrations in sediment comprises an important focus of environmental impact/quality monitoring programmes in many regions of the world. In addition to environmental concerns there are several pragmatic reasons for monitoring contaminant concentrations in sediment. The higher concentrations of chemicals/contaminants in sediment compared to the overlying water column make detection and measurement in the laboratory easier. Furthermore, the low and often highly variable concentrations of contaminants in the water column due to variation in turbulence/mixing and variable anthropogenic inputs means that only a snapshot of water quality status is gained and important contamination events may be missed. Analysis of sediment provides a far more conservative, spatially and temporally integrated measure of contamination events and problems.

The contaminant status of sediment thus provides important information for assessing environmental quality, and in the context of the survey discussed in this report, for assessing the environmental impact of effluent discharge. The accumulation of contaminants in sediment near outfalls is dependent on several factors. These include the proportion of fine-grained material (especially mud) on the seabed, the concentration and type of contaminants in the effluent, and the dispersal/deposition of potentially contaminated particulate material in the effluent.

This chapter analyses and discusses findings of the analysis of various physical, chemical and biological variables in sediment collected on 24 and 25 May 2011. The major objectives are to assess the magnitude and spatial extent of effluent discharge on sediment quality, estimate the potential biological significance of chemical concentrations in sediment, and analyse temporal trends in sediment quality.

4.2. Materials and Methods

4.2.1. Fieldwork

A Day grab that sampled a surface area of 0.25 m^2 was used to collect sediment from 15 stations arranged in a grid-like manner spanning the diffuser sections of the Central Works and Southern Works outfalls (Figure 4.1). The stations cover an area of about 0.32 km^2, with stations situated about 200 m apart in offshore and alongshore directions. The most distant stations of each grid (B2, B3 and B4 in Figure 4.1) are situated 500 m from the diffuser section of the outfalls. Three additional sediment samples were collected at each station along the central line of the grid running parallel to the shoreline (the so-called 3 line), to provide a total of four replicate samples. Sediment from three of the

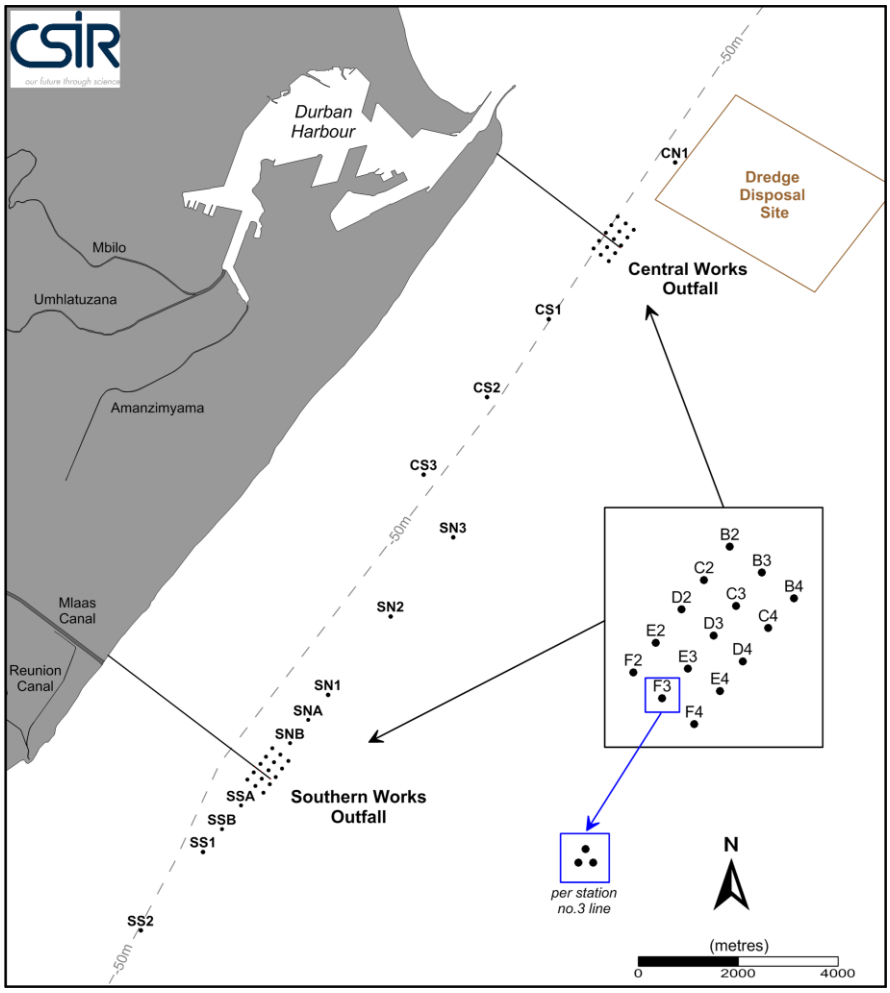

Figure 4.1. Map illustrating the positions where sediment for physical and chemical analysis was collected for the 2011 survey of the Durban outfalls monitoring programme.

samples was used for grain size, total organic content, metal and benthic invertebrate community analysis. The fourth sample provided sediment for organic chemical analysis and microbiological and toxicity testing. Four sediment samples were also collected at each of the 13 reference sites situated to the north-northeast and south-southwest of the outfalls. There were four reference sites for the Central Works outfall and nine for the Southern Works outfall (Figure 4.1). A total of 112 successful grab samples was required to satisfy the sampling design.

For discussion purposes, stations of the grid at which replicate sediment samples were collected are referred to as sites, while the entire grids, either including or excluding non-replicated stations, are referred to as the Central Works or Southern Works outfall sites.

The Day grab used to collect sediment has hinged top screens that prevent disturbance and washout of sediment during retrieval. The screens also permit the removal of undisturbed surface sediment from the grab. Following retrieval of the grab an inspection of the contents was made. Undisturbed and level sediment inside the grab was required for the grab to be accepted. If these conditions were not met the contents were discarded and the grab was again deployed. For successful grabs, water overlaying sediment was siphoned off, the contents photographed and inspected for texture, colour, aroma and the presence of obviously anomalous matter (e.g. tomato seeds, cigarette butts). The findings of these observations were noted on field data sheets. For most grabs approximately 150 cm^3 of surface sediment was removed and homogenised in a stainless steel bowl using a stainless steel spoon. Two approximately 50 g and one 30 g aliquots of sediment were transferred from the bowl to pre-cleaned high density polyethylene containers and stored on ice until return to the laboratory, where the sediment was frozen (-4^{o}C) until analysis. These samples were for metal, grain size, total organic

content and chemical oxygen demand analysis. The remainder of the sediment was then sieved to extract benthic macrofauna (see Chapter 5). As mentioned above, the grab was deployed a fourth time at reference sites and at sites on the outfall grid. Approximately 1 kg of surface sediment from the fourth grab was transferred to Ziploc® plastic bags for toxicity testing, 200 g to hexane rinsed glass bottles for organic chemical analysis, 100 g to autoclaved glass bottles for microbiological analyses, and about 50 g to high density polyethylene containers for grain size and total organic content analysis. These samples were also stored on ice until return to the laboratory, where the sediment was either refrigerated (4^{o}C, toxicity testing), frozen (organic chemical analysis), or analysed immediately (microbiology).

The sampling vessel steamed to position each time the grab was deployed at a site where replicate samples were collected or where the grab contents were deemed unacceptable. The grab was rinsed with site water and scrubbed with a hard brush before each deployment.

4.2.2. Laboratory Analyses

4.2.2.1. Accredited Laboratory

Analyses were performed in environmental chemistry laboratories at CSIR campuses in Durban and Stellenbosch. These are the only laboratories in South Africa that are accredited by the South African National Accreditation System (SANAS) for the analysis of marine water, sediment and tissue samples.

4.2.2.2. Faecal Indicator Bacteria

Approximately 50 g of wet sediment was weighed into autoclaved glass bottles and 150 ml of sterilised saline solution added. The sediment/saline solution was then shaken vigorously on an automated shaker. The solution was allowed to stand for about five minutes to permit fine-grained material in suspension to settle. The supernatant was removed and treated as described below. An aliquot of sediment was also removed and dried in an oven at 105^{o}C for 24 hrs, to permit the expression of bacterial counts (colony forming units, cfu) per dry weight of sediment.

The supernatant was analysed for faecal coliforms and faecal streptococci by membrane filtration techniques as described in APHA (1995). This involved passing an aliquot of the sample or a volume of an appropriate dilution through a 47 mm diameter sterile membrane filter with a 0.45 µm pore size. The membranes were then placed on selective media and incubated and analysed following methods outlined by ISO 7899/2 (1984), Mates and Shaffer (1988), SABS (1990) and APHA (1995).

4.2.2.3. Sediment Grain Size Composition

The grain size composition of sediment was determined by wet and dry sieving into seven grain size classes according to the Wentworth Scale, namely: mud (<0.063 mm), very fine-grained sand (0.063 - 0.125 mm), fine-grained sand (0.125 - 0.250 mm), medium-grained sand (0.25 - 0.50 mm), coarse-grained sand (0.5 - 1.0 mm), very coarse-grained sand (1.0 - 2.0 mm) and gravel (>2.0 mm). Grain size classes are expressed as a fraction of bulk sediment dry weight.

4.2.2.4. Total Organic Content

Sediment samples were oven dried, weighed, and organic debris then degraded using hydrogen peroxide. The sediment was washed in distilled water, re-dried and re-weighed, and the difference in dry weight before and after degradation used to determine the total organic content. Total organic content is expressed as a fraction of sample dry weight.

4.2.2.5. Chemical Oxygen Demand

Sediment was thawed, homogenised, and potassium permanganate, sodium hydroxide and manganese sulphate added to weighed samples and incubated in a water bath. Chemical oxygen demand was then determined titrimetrically, using sodium thiosulphate with starch as an indicator. Chemical oxygen demand is expressed on a mg $O_2.g^{-1}$ basis.

4.2.2.6. Metals

Sediment was thawed, homogenised, and approximately 1 g weighed into a high-pressure digestion vessel. Sediment was then digested in concentrated nitric acid (HNO_3), with microwave assistance. Digestates were diluted to volume with deionised water and concentrations of various major

and minor elements detected and quantified using Inductively Coupled Plasma Optical Emission Spectroscopy. Sediment moisture content was determined by drying a similar mass of wet sediment to that mentioned above in an oven at 105°C for 24 hrs. Metal concentrations are expressed on a dry weight basis, as $mg.g^{-1}$ or $\mu g.g^{-1}$.

4.2.2.7. Organic Chemicals

A suite or organic chemicals, including persistent organic pollutants, was analysed. These included seven polychlorinated biphenyl congeners (so-called ICES 7), 16 polycyclic aromatic hydrocarbon isomers, and 31 parent or metabolite organochlorine pesticides. Organic chemicals were extracted from sediment using organic solvents. Extracts were further processed to remove interfering substances and to concentrate analytes. Concentrations of analytes were then identified and confirmed using a gas chromatograph-mass spectrometer or high performance liquid chromatography. Quantification was relative to external standards. Organic chemical concentrations are expressed on a dry weight basis, as $\mu g.kg^{-1}$.

4.2.2.8. Porewater Toxicity Testing

The toxicity of the porewater of sediment collected at one of the replicate sediment samples from each site was measured in the laboratory using the sea urchin fertilisation test (see Chapter 2, section 2.2.2 for a description of the toxicity test procedure). Porewater was extracted from sediment through centrifuging.

To provide an indication of whether ammonia, which is highly toxic to marine organisms at elevated concentrations (specifically in the form NH_3-N) (DWAF 1995), could explain trends in toxicity, where sufficient porewater was available this was analysed for total ammonia-N using the same method presented in a previous chapter (see Chapter 3, section 3.2.2.4).

4.2.3. Data Analysis

Sediment grain size is reported using the Wentworth Scale (Table 4.1). Various sediment grain size statistics were determined, including the mean and median particle diameter, sorting coefficient, and the dry weight fraction of the bulk sediment comprised by each grain size class. Sediment texture is classified according to Shepard (1954). The data are presented in the form of a ternary (sometimes called triangle) plot, which permits the presentation of three variables in a two-dimensional plot.

Average values for some variables were compared between outfall and reference sites using one-way Analysis of Variance (ANOVA), with inferences at the $\alpha = 0.05$ level. A post-hoc Student-Newman-Keuls test was used to identify homogenous groups when statistically significant differences were detected.

The strength of linear relationships between certain physical and chemical variables was assessed through correlation and linear regression analysis. Correlation analysis determines whether changes in two or more variables are positively or negatively related and the strength of the relationship. Linear regression also determines whether changes in two or more variables are positively or negatively related and the strength of the relationship, but here change in one of the variables is considered dependent on change in the other variable. Thus, a linear regression provides predictive ability.

The proportion of the variability associated with the correlation between two variables is measured by the correlation coefficient (r value), or in the case of linear regression analysis by the coefficient of determination (r^2 value). The correlation coefficient

Table 4.1. A subset of the Wentworth scale representative of sediment encountered in the area monitored. Sorting categories are based on those defined by Folk (1966).

Grain Size (mm)	Description	Standard Deviation	Sorting Category
2	Granule/Gravel	<0.35	Very well sorted
1	Very coarse grained sand	0.35-0.50	Well sorted
0.5	Coarse grained sand	0.50-0.71	Moderately well sorted
0.25	Medium grained sand	0.71-1.00	Moderately sorted
0.125	Fine grained sand	1.00-2.00	Poorly sorted
0.0625	Very fine grained sand	>2.0	Very poorly sorted
<0.0625	Mud (silt and clay)		

and coefficient of determination can range in value between zero and one - a value of zero indicates that there is no linear relationship between the variables and a value of one indicates a perfectly linear relationship. A negative value for the correlation coefficient or the slope of a regression indicates that there is an inverse relationship between the variables. Of additional importance is the statistical significance of the correlation or regression. Although a correlation or regression may be statistically significant, this does not necessarily mean that it is meaningful. Weak relationships (i.e. low r or r^2 values), regardless of their statistical significance, have limited predictive value. For example, if the coefficient of determination is 0.30 and is statistically significant (at, for example, α = 0.05), little importance should be accorded the regression since a very substantial amount of the variability between the two variables has nothing to do with the correlation between them (Goodsell et al. 2009). Results from and usability of correlation analyses were confirmed by graphical analysis of data.

The contaminant status of metal concentrations in sediment was interpreted using baseline metal concentration models and baseline metal concentrations defined for sediment from KwaZulu-Natal coastal waters by the Coastal Systems research group of the CSIR. The baseline models and baseline concentrations are the only available tool for reliably determining whether a sediment sample is metal enriched/contaminated. A description of the procedures used to define the baseline models and baseline concentrations is beyond the scope of this report. However, a description of baseline model and baseline concentration application for

Table 4.2. Sediment quality guidelines defined by the Department of Environmental Affairs and by Long et al. (1995). Metals in $\mu g.g^{-1}$ and organic contaminants in $\mu g.kg^{-1}$ dry weight.

Chemical class	Chemical	Department of Environmental Affairs		Long et al. (1995)	
		Action Level/ Special Care Level	Prohibition Level	Effects Range Low	Effects Range Median
Metals	Arsenic	30	150	8.2	70
	Cadmium	1.5	10	1.2	9.6
	Chromium	50	500	81	370
	Copper	50	500	34	270
	Lead	100	500	46.7	218
	Mercury	0.5	5	0.15	0.71
	Nickel	50	500	20.9	51.6
	Zinc	150	750	150	410
Organic chemicals	Oils	1000	1500	-	-
	Acenaphthene	-	-	16	500
	Acenaphthylene	-	-	44	640
	Anthracene	-	-	85.3	1100
	Fluorene	-	-	19	540
	2-Methylnaphthalene	-	-	70	670
	Naphthalene	-	-	160	2100
	Phenanthrene	-	-	240	1500
	ΣLow-molecular weight PAH	-	-	552	3160
	Benz(a)anthracene	-	-	261	1600
	Benzo(a)pyrene	-	-	430	1600
	Chrysene	-	-	384	2800
	Dibenzo(a,h)anthracene	-	-	63.4	260
	Fluoranthene	-	-	600	5100
	Pyrene	-	-	665	2600
	ΣHigh molecular weight PAH	-	-	1700	9600
	Total PAH	-	-	4022	44792
	Pesticides	-	500	-	-
	p,p'-DDE	-	-	2.2	27
	Total DDT	-	-	1.58	46.1
	Total PCBs	-	-	22.7	180

interpreting metal concentrations measured in sediment is provided in the Results and Discussion section of this chapter. Similar procedures were used to define baseline models for total organic content and chemical oxygen demand, which relationships were then used to identify anomalous measurements and the magnitude of the anomalies in a similar manner to the metal baseline models. Further details are provided in subsequent sections of this chapter.

The potential biological significance of metal concentrations in sediment was assessed relative to sediment quality guidelines derived by Long et al. (1995) for application in North American coastal waters, and to sediment quality guidelines defined by the Department of Environmental Affairs for South African coastal waters (undated document; see Table 4.1).

4.3. Results and Discussion

4.3.1. Faecal Indicator Bacteria

The usefulness of faecal indicator bacteria as a tool for tracing effluent dispersion in the water column (see Chapter 3) extends to sediment. The general opinion is that since faecal indicator bacteria tend to attach to particulate (organic) matter, their presence in sediment reflects the recent settling of effluent derived particulate matter on the seabed. The situation is more complex, however, since numerous studies have identified sediment and sand on the shoreline as important reservoirs for faecal indicator bacteria in estuarine and marine environments (e.g. Ferguson et al. 2005, Bevesdorf et al. 2006, Yamahara et al. 2007). The bacteria may persist for long periods in sand and sediment and may multiply under suitable conditions, even in the absence of persistent faecal contamination (e.g. Gerba and McLeod 1976, LaLiberte and Grimes 1982, Davies et al. 1995). Certainly, it would appear that faecal indicator bacteria in sediment show enhanced survival compared to the same bacteria in the water column (Craig et al. 2004). Most recent studies in this context have focussed on beach sand as a potential source of bacterial impairment of bathing waters (e.g. Whitman and Nevers 2003, Ferguson et al. 2005, Beversdorf et al. 2006, Bonilla et al. 2007, Yamahara 2007). In most studies, faecal indicator bacteria counts in sand and/or sediment were higher than in the adjacent or overlying water column, often by several orders of magnitude. Thus, while the presence of faecal indicator bacteria in sediment around deepwater effluent outfalls may not necessarily indicate recent faecal contamination, their presence does provide an understanding of the spatial extent of effluent dispersal and influence on the benthic environment.

Faecal indicator bacteria were detected in sediment at all sites (Figure 4.2). The faecal coliform bacteria colony forming unit count at reference site CN1, situated 2000 m to the northeast of the Central Works outfall, was low (63 colony forming units). Colony forming unit counts were markedly higher at sites nearer the outfall, peaking at 43 308 colony forming units at site CWC3, situated 260 m to the northeast of the outfall. Counts decreased sharply at sites CWD3 and CWE3, to the southwest of the outfall, and remained relatively low and generally comparable through to reference site CS2. The count then increased at reference site CS3. The trend for faecal streptococci bacteria was different since the highest colony forming unit count was at site CWF3 (1620 colony forming units), situated 360 m to the southwest of the outfall. The count at reference site CS1 (1340 colony forming units), situated 2000 m to the south-southwest of the outfall, was only slightly lower compared to site CWF3. There was no particularly pronounced trend for the remaining sites.

For the Southern Works outfall, faecal coliform bacteria colony forming unit counts at numerous reference sites to the northeast of the outfall were comparable to counts at sites situated within 300 m to the northeast of the outfall (Figure 4.2). The highest count (22 682 colony forming units) was at site SWE3, situated 100 m to the southwest of the outfall. Counts decreased sharply at site SWF3 and remained relatively low at reference sites situated to the southwest of the outfall. With the exception of site SWE3, faecal coliform bacteria colony forming unit counts at sites to the southwest of the outfall were usually substantially lower compared to sites to the northeast of the outfall. A broadly comparable trend was evident for faecal streptococci bacteria. The implication is that effluent was being dispersed predominantly in a northeasterly direction.

Faecal indicator bacteria colony forming counts were

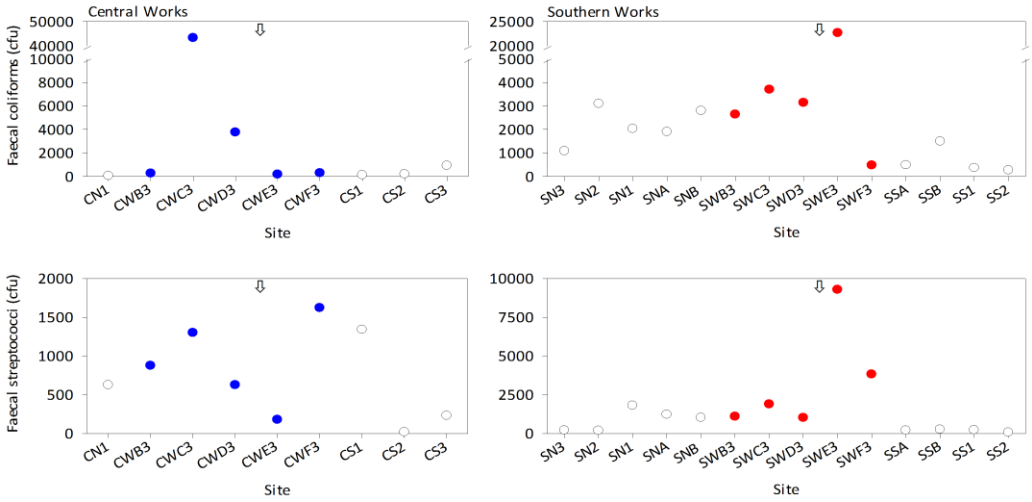

Figure 4.2. Faecal indicator bacteria colony forming unit (cfu) counts in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

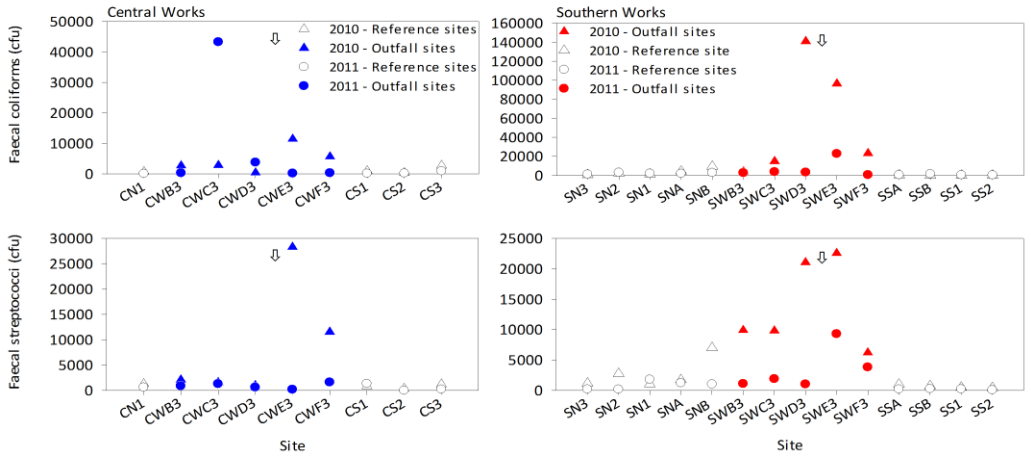

Figure 4.3. Comparison of faecal indicator bacteria colony forming unit (cfu) counts in sediment collected for the 2010 and 2011 surveys of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Note breaks in the y-axis for two plots.

generally higher in sediment from Southern Works outfall and reference sites compared to Central Works outfall and reference sites (Figure 4.2). This is similar to the findings for the 2010 survey. On a weight basis (i.e. considering 100 g = 100 ml), faecal indicator bacteria colony forming unit counts in sediment were far higher (up to several orders of magnitude) compared to counts in the water column (see Chapter 3).

It is only possible to compare trends for faecal indicator bacteria in sediment between the 2010 and 2011 surveys since these are the only surveys for which this monitoring was performed. Faecal coliform counts in 2010 and 2011 were generally highest at outfall sites, and the highest count was always at an outfall site (Figure 4.3). Although this trend was also applicable to faecal streptococci at the Southern Works outfall, counts were substantially higher in 2010 compared to 2011 for the Central Works outfall. The implication is that the most significant settlement of effluent derived particulate material is occurring on the seabed in the immediate vicinities (i.e. within about 500 m) of the outfall diffusers. Nevertheless, the presence of faecal indicator bacteria in sediment as far as 6000 m from the outfalls, which is the furthest distance monitored, implies that effluent derived particulate material is settling on the seabed across the entire study area. This is important since, technically, it negates use of the reference sites as points for comparison to outfall sites, since all reference sites

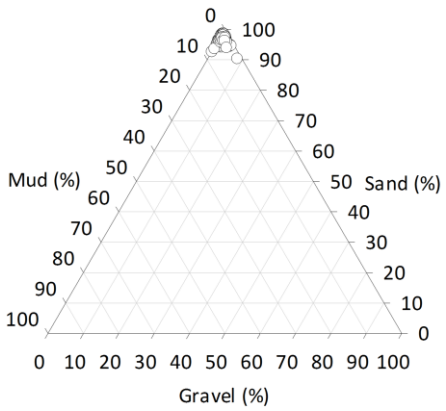

Figure 4.4. Ternary plot illustrating the proportional contribution of gravel, sand and mud to bulk sediment collected for the 2011 survey of the Durban outfalls monitoring programme.

are impacted by the effluent. This said, as discussed in subsequent sections of this chapter and in Chapter 5, other indicators of environmental condition did not provide evidence for significant adverse impacts to the benthic environment at the majority of reference sites.

Although there seems little doubt the faecal indicator bacteria in sediment at reference sites was derived from effluent, it is uncertain whether this reflects recent faecal matter contamination or bacteria persisting and perhaps multiplying in the sediment. It is also uncertain whether effluent discharge has led to elevated faecal indicator bacteria counts in sediment closer to the shoreline since this was not an objective of the monitoring programme. However, considering that faecal indicator bacteria colony forming counts in water samples were low (see Chapter 2) and historically have been low in surface waters near and distant to the outfalls (e.g. CSIR 2010, 2011), the probability that effluent discharge is contributing to microbiological impairment of shoreline waters and thereby endangering human health seems remote.

4.3.2. Sediment Grain Size Composition

An understanding of the grain size composition of sediment provides important information for understanding factors that influence the composition and structure of benthic macrofaunal communities. The grain size composition of sediment also provides information on the amount of wave action, current velocities and habitat stability in a particular area, which also affects benthic macrofaunal communities (this is addressed in Chapter 5). The grain size composition of sediment also provides information for interpreting trends in contaminant concentrations. This is because fine-grained sediment, such as silt and clay, sequesters more contaminants than coarse-grained sediment (e.g. sand), because of the greater surface area to volume ratio provided by the fine particles for contaminant adsorption and since the surface of the grains are electrically charged and this render them more chemically reactive and facilitates contaminant/chemical adsorption (Plumb 1981, Power and Chapman 1995). Contaminant concentrations tend, therefore, to be highest in fine-grained and lowest in coarse-grained sediment. In the context of outfall monitoring programmes, effluent discharge has also been shown to alter the grain size composition of sediment through the input of significant amounts of particulate organic material (e.g. Read et al. 1989, Gray 1997, Arvai et al. 2002). Lastly, the presence of outfalls and associated ballast material can alter the local hydrodynamic regime and thus influence the grain size composition of sediment in the vicinity of the outfall, due to changes in current velocities that affect the bedload transport of sediment.

The sediment at all stations/sites was dominated by sand (average sand fraction = 96.92%, 5^{th} percentile = 94.22%, 95^{th} percentile = 98.41%; Figure 4.4). Medium-grained sand was always the dominant grain size class at Southern Works outfall and reference sites (Figure 4.5). For Central Works outfall and reference sites, either medium- or fine-grained sand was the dominant grain size class, with each size class generally similarly represented (Figure 4.5). Apart from this difference in the dominant grain size class between the outfall lines of sites, the contribution of very coarse- and coarse-grained sand at Southern Works outfall and reference sites was generally higher compared to Central Works outfall and reference sites (Figure 4.5). The differences allude to differences in current regimes at the depths over which the diffusers for each outfall are positioned (see Table 1.1).

Mud was poorly represented across the study area (average sand fraction = 2.16%, 5^{th} percentile = 1.02%, 95^{th} percentile = 3.93%; Figures 4.4 and 4.5).

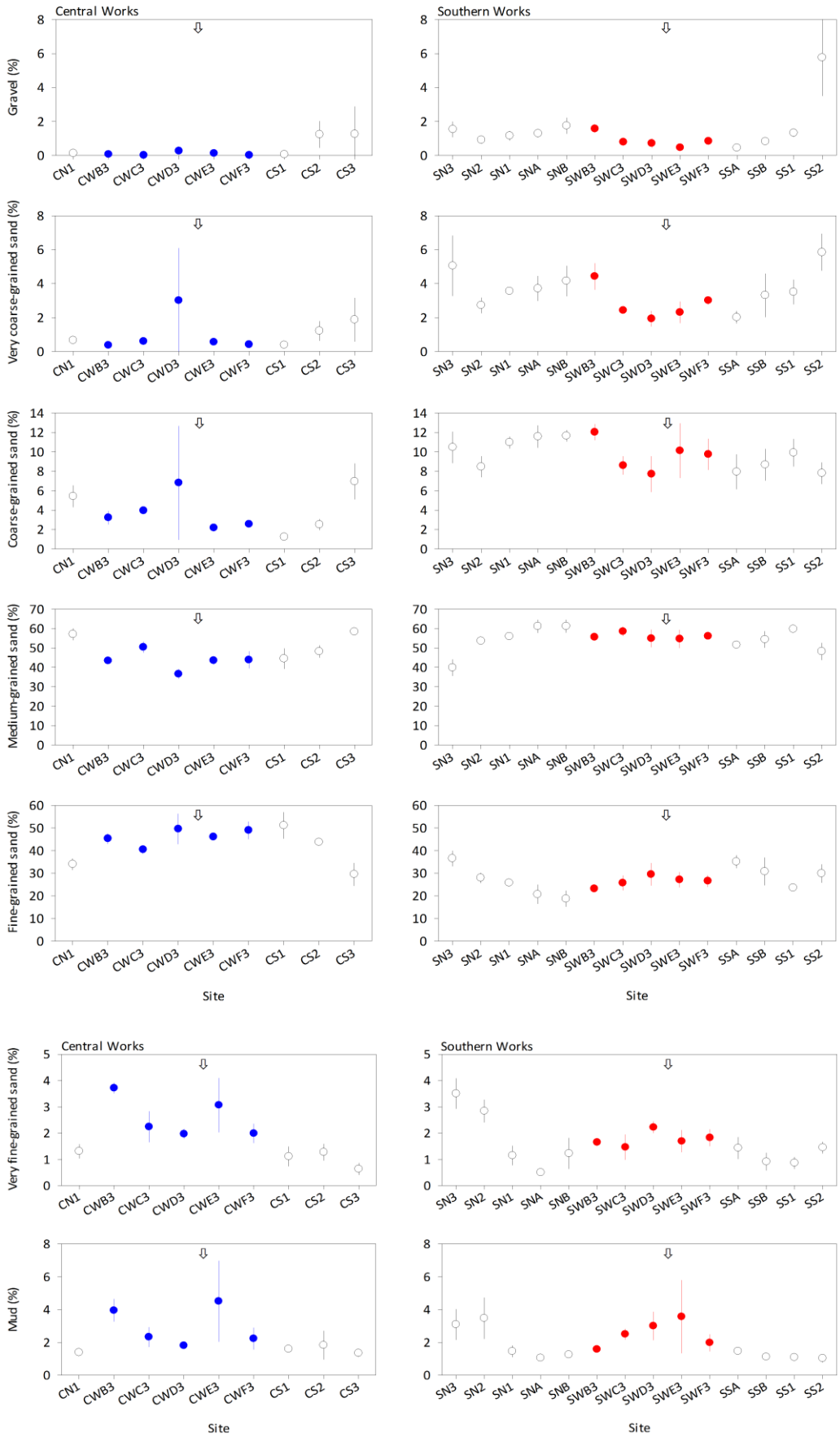

Figure 4.5. Average (± one standard deviation) contribution of grain size classes to bulk sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

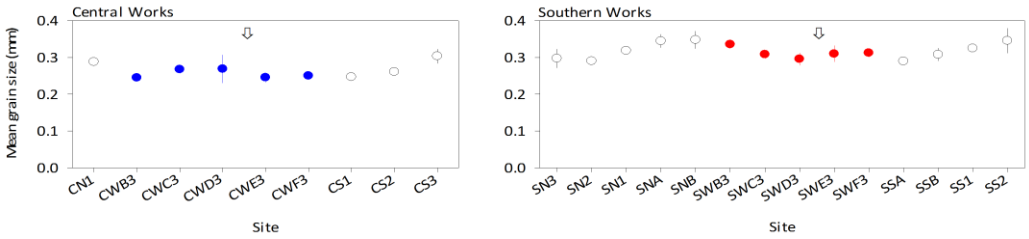

Figure 4.6. Average (± one standard deviation) grain size of sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

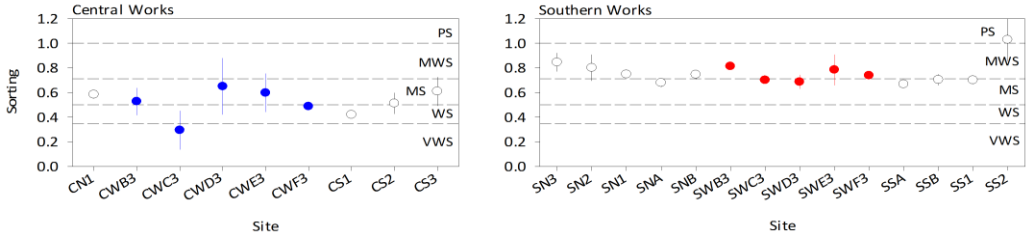

Figure 4.7. Average (± one standard deviation) sorting coefficients of sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites. PS = poorly sorted, MWS = moderately well sorted, MS = moderately well sorted, WS = well sorted, well sorted, VWS = very well sorted.

The mud fraction of sediment for Central Works outfall and reference sites was variable, but tended to be slightly to somewhat higher in the vicinity of the outfall compared to reference sites (Figure 4.5). For the Southern Works outfall, the mud fraction at reference sites SN2 and SN3 was amongst the highest. The contribution of mud to bulk sediment then decreased at reference site SN1, SNA and SNB to the northeast of the outfall, peaked at stations either side of the diffuser before decreasing with increasing distance to the southwest of the outfall. The elevated mud fraction of sediment at sites SN2 and SN3 was also evident in the 2009 and 2010 surveys of the Durban outfalls monitoring programme (CSIR 2010, 2011) and alludes to an oceanographic feature that facilitates the settling of fine-grained material on the seabed in this area. Nevertheless, the mud fraction at these sites was low (highest contribution of 4.87%) compared to sheltered coastal habitats, such as estuaries.

The average grain size at Central Works outfall and reference sites was generally slightly smaller compared to Southern Works outfall and reference sites (Figure 4.6). This alludes to a difference in the current regime at the different depths over which the diffusers for each outfall are positioned.

Sorting refers to the range of grain sizes represented in sediment. Sediment that is well sorted is comprised predominantly of grains of roughly the same size while poorly sorted sediment has a wide range of grain sizes. Well-sorted sediment is characteristic of high-energy environments, where currents, waves and other forms of turbulence are of sufficient velocity to winnow fine grains (e.g. mud) from the sediment, but not coarser grains. Poorly sorted sediment is characteristic of low energy environments (e.g. estuaries), where turbulence velocity is insufficient to winnow even very fine grains of sediment. Poorly sorted sediment may also reflect recent sediment dumping (e.g. after floods), which in time may be sorted.

Sediment at Central Works outfall and reference sites tended to be better sorted compared to Southern Works outfall and reference sites (Figure 4.7). The difference again alludes to a difference in the current regime at the different depths over which the diffusers for each outfall are positioned. The implication is that currents are of a lower average velocity at the depth that the Southern

Works outfall and reference sites are situated compared to the Central Works outfall and reference sites. This has implications then in terms of effluent dispersion and the bedload transport of sediment-sorbed contaminants.

Perhaps the most significant finding arising from analysis of the grain size composition of sediment is that the study area is not of a depositional nature. In other words, prevailing currents are of a sufficient strength to prohibit the settling and accumulation of 'significant' volumes of fine-grained material. Nevertheless, there is clear evidence that effluent derived particulate material is settling on the seabed in the vicinities of the outfalls, as indicated by trends in sediment faecal indicator bacteria counts (see above) and total organic content (see below).

4.3.3. Total Organic Content

Total organic content provides a measure of the amount of (generally particulate) organic matter in sediment and is an important indicator of effluent discharge impacts, particularly for sanitary effluent which is typically rich in particulate organic matter. Organic matter in surficial sediment comprises an important source of food for many benthic and epibenthic fauna (e.g. deposit feeding polychaetes; see for example Pearson and Rosenberg 1978, Lopez and Levinton 1987, Snelgrove and Butman 1994, Hyland et al. 2005). However, when the deposition of organic matter exceeds the rate at which consumers can process this material, it may lead to the development of hypoxia and in extreme situations anoxia in sediment and bottom waters when oxygen-consuming bacteria break down the matter and consume oxygen at a rate greater than the rate of re-ventilation. This can lead to the wholesale disturbance of benthic macrofaunal communities and have ripple-like impacts through ecosystems (Pearson and Rosenberg 1978, Diaz and Rosenberg 1995, Gray et al. 2002). An understanding of the total organic content of sediment has further implications in the context of the Durban outfalls monitoring programme since it provides a surface for (particularly organic) contaminant adsorption and retention.

There are no prescriptive values to define when the total organic content of sediment can be regarded as low, medium or high, since the content is unique to different environments. Sediment from sheltered estuarine environments, for example, tends to naturally have a higher total organic content compared to marine nearshore waters, where greater turbulence and generally stronger currents restrict the settling and accumulation of this fine-grained material on sediment.

The total organic content in sediment on the grid of stations spanning the Central Works outfall was, on average, slightly higher compared to sediment at the reference sites (0.48 ± 0.17% and 0.28 ± 0.08% respectively; this difference was statistically highly significant, one-way ANOVA, $p = 0.001$). At the site specific level, the average total organic content was highest at site CWB3, which was situated 500 m to the northeast of the diffuser, and then generally progressively decreased to site CWF3 (Figure 4.8).

The average total organic content in sediment at the grid of stations spanning the Southern Works outfall was also slightly higher compared to sediment at the reference sites (0.46 ± 0.18% and 0.24 ± 0.11% respectively; this difference was statistically highly significant, one-way ANOVA, $p < 0.001$). The average total organic content at reference sites SN3 and SN2 was relatively high, and then decreased in the direction of reference site SNA (Figure 4.7). The average total organic content then progressively increased as the outfall was approached and peaked at sites situated to the immediate northeast and southwest of the diffuser (i.e. sites SWD3 and SWE3), and then progressively decreased with increasing distance to the southwest of the outfall (Figure 4.8).

The abovementioned trends attest to a generally higher total organic content in sediment in the immediate vicinities of the outfalls compared to the reference sites. However, the direct comparison of total organic content between stations/sites can be misleading since in anthropogenically un-impacted areas the mud fraction and the total organic content of sediment is usually strongly positively correlated. The is because these fine-grained materials are similarly winnowed from or deposited on sediment depending on prevailing currents and they tend, therefore, to accumulate in or be depleted from the same areas. Thus, sand dominated sediment naturally has a lower total organic content than mud

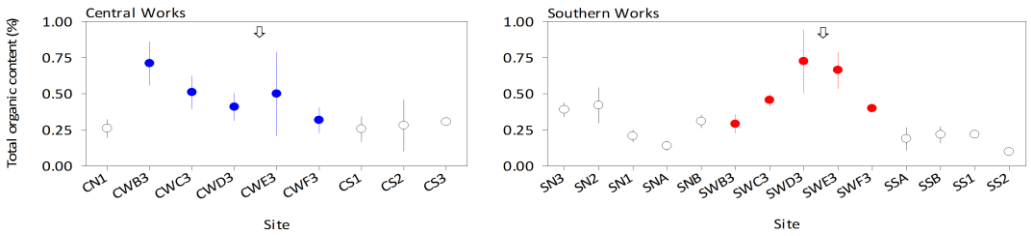

Figure 4.8. Average (± one standard deviation) total organic content of sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

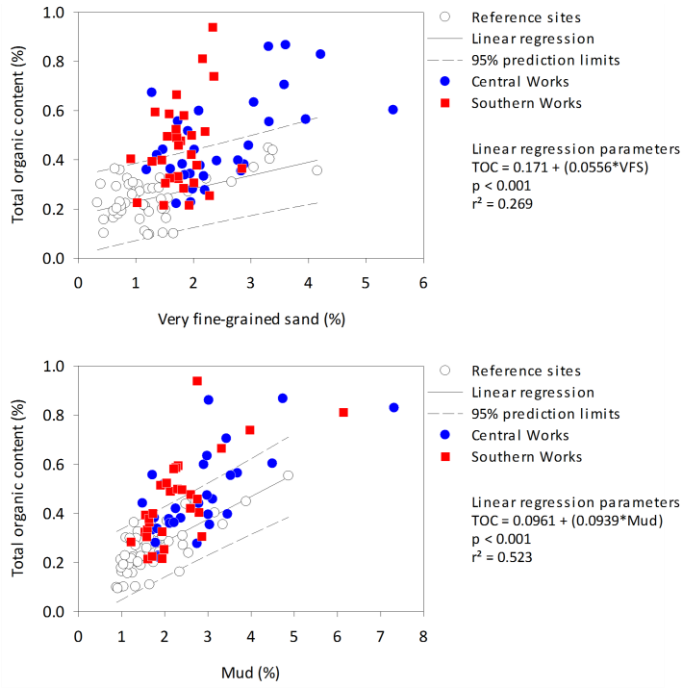

Figure 4.9. Relationship between total organic content and the very fine-grained sand and mud fraction of sediment collected for the 2011 survey of the Durban outfalls monitoring programme.

dominated sediment. The influence of sediment grain size on total organic content must thus be compensated for before a direct comparison can be made between stations/sites. The natural relationship between the mud fraction and total organic content of sediment can be modelled through linear regression analysis, and the regression and its associated prediction limits can be used to interpret whether sediment from (putatively) anthropogenically impacted areas is enriched with organic matter.

The relationship between the mud fraction and the total organic content of sediment at reference sites was weak in surveys performed in 2009 and 2010. Therefore, the stronger relationship between the very fine-grained sand fraction and total organic content was used to develop an interpretive tool for those surveys. The situation was different for the 2011 survey, since the relationship between total organic content and mud was stronger than for very fine-grained sand (Figure 4.9). Consequently, both relationships are used to develop an understanding of whether the total organic content of sediment from stations/sites in the vicinities of the outfalls is higher than expected based on data for the reference sites, and to allow comparison to the findings of previous surveys. In Figure 4.9, the solid regression line represents the average total organic content at the reference sites at any particular very fine-grained sand or mud fraction. The dashed lines flanking the regression line are the 95% prediction limits, which represent the range within which the total organic content in 95% of 'un-impacted' sediment samples is expected to fall.

Figure 4.10. Total organic content Enrichment Factors determined from regressions using very fine-grained sand (top) and mud (bottom) as the normaliser. Enrichment Factors >1 (horizontal dashed lines) indicate that the total organic content was higher than the range predicted for sediment from reference sites. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

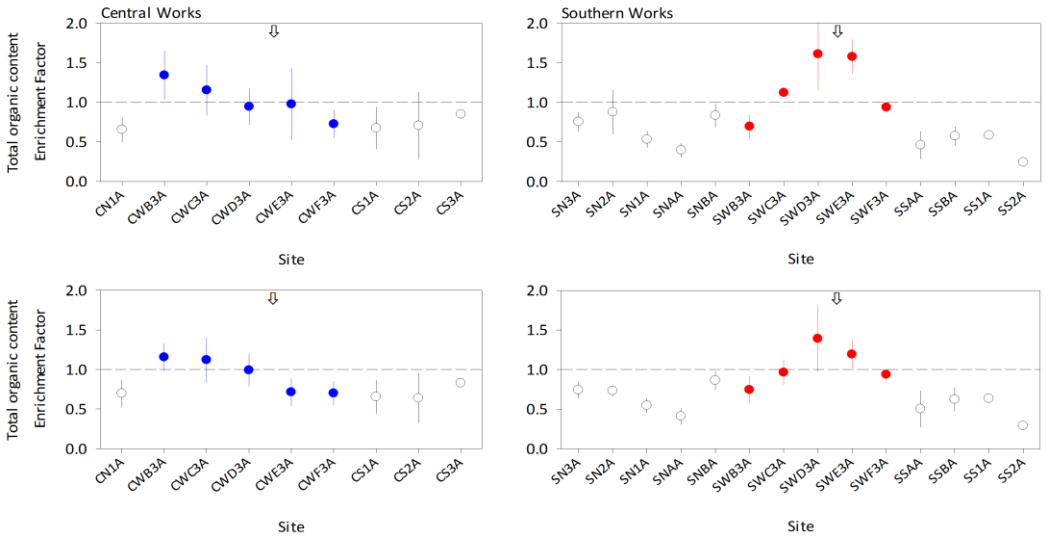

Figure 4.11. Average (± one standard deviation) total organic content Enrichment Factors determined from regressions using very fine-grained sand (top) and mud (bottom) as the normaliser. Enrichment Factors >1 (horizontal dashed lines) indicate that the total organic content was higher than the expected range for sediment from reference sites. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

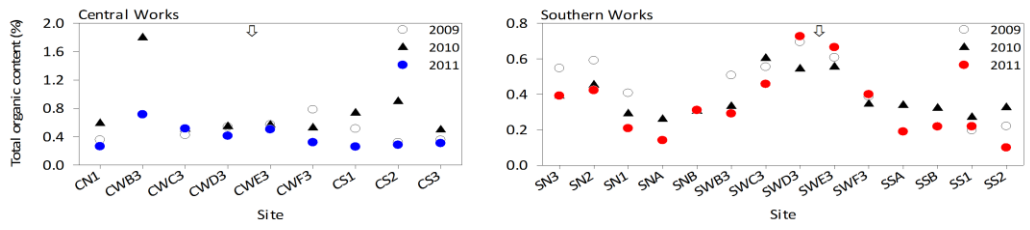

Figure 4.12. Comparison of the average total organic content of sediment for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. Error bars are not included for presentation purposes. Arrows denote the positions of sites relative to the outfall diffusers.

Because the strengths of the regressions are weak to very weak (as denoted by the low coefficients of determination, or r^2 values) they have little predictive power, but can nevertheless be used to identify samples with an anomalous total organic content. Superimposed on the relationships are data for the outfall stations/sites. Data points that fall above the upper prediction limit (i.e. upper dashed line) represent sediment that was enriched with organic matter relative to the reference sites. Data that fall between the prediction limits is within the range for reference sites. It is important to note that some reference sites may be enriched with particulate organic matter, as none of the reference sites can be regarded as truly of a background nature considering their proximity to the outfalls and particularly due to the fact that faecal indicator bacteria were present in sediment at all reference sites (see section 4.3.1 of this chapter). Nevertheless, the incidence of enrichment at reference sites is believed to be small, with the result that the baseline relationship can be used as an interpretive tool.

Using simple mathematics based on the assumption that the 95% prediction limits are essentially linear (they are in fact biconcave, the deviation from linearity is small), the difference between the upper prediction limit and total organic content in each sediment sample was calculated to define an Enrichment Factor (Figures 4.10 and 4.11). An Enrichment Factor >1 means that the total organic content exceeds the upper prediction limit, that is, falls above of the upper bound for the reference sites. As is evident from Figures 4.10 and 4.11, only sediment in the vicinities the outfalls was enriched with organic matter to any significant degree. Although qualitatively similar, the actual number of samples interpreted as enriched with particulate organic matter and the magnitude of the enrichment differed depending on whether very fine-grained sand or mud was used as the normaliser (Figure 4.10). The average Enrichment Factor for total organic content at reference site SNB, situated 100 m to the northeast of the outfall, was only slightly lower than one (Figure 4.11) and suggests this material was derived from effluent. If so, then particulate organic material is accumulating in sediment up to 1000 m from the outfall diffuser to a degree higher than at the reference sites.

The trend for average total organic content in sediment at Central Works and Southern Works outfall sites is similar to that observed in the 2009 and 2010 surveys for the Durban outfalls monitoring programme, although the absolute contribution varied slightly between surveys (Figure 4.12). In other words, there was a consistent pattern of enrichment of sediment with particulate organic matter between 2009 and 2011.

A notable trend also evident in the 2009 and 2010 surveys was the increase in total organic content in sediment between reference sites SN1 and SN3. As discussed previously, a comparable trend for the mud fraction of sediment attests to an oceanographic feature that facilitates the settling of fine-grained material on the seabed in the vicinity of sites SN2 and SN3, but whether this material is derived (predominantly) from effluent or other sources is uncertain.

Although there are numerous sources of organic matter to sediment in marine aquatic ecosystems, considering that sediment in the vicinities of the outfalls was usually the most enriched with organic matter in the 2011 survey, that this trend has been persistent between 2009 and 2011[6], and that effluent is typically rich in particulate organic matter, there is strong evidence that the excess organic matter in sediment in the vicinities of both outfalls was derived from effluent. This is supported by the generally higher faecal indicator bacteria colony forming unit counts in sediment in the vicinities of the outfalls compared to reference sites. The trend for faecal indicator bacteria in sediment does not provide compelling evidence that organic matter in sediment at reference sites SN2 and SN3 is derived from effluent, although these bacteria were present in the sediment and attests to the fact that effluent derived material is impinging on these sites.

As stated previously, there are no prescriptive values to define when the total organic content of sediment can be regarded as low, medium or high, since the content is unique to different environments. It is, therefore, not possible to infer directly whether organic matter in sediment is adversely impacting

[6] The enrichment of sediment in the vicinities of the outfalls with organic matter was also evident prior to 2009 but was not as easily demonstrated due to the sampling design used at the time.

benthic invertebrate communities. There is, however, strong indirect evidence for such an effect from analyses of benthic macrofaunal communities in the vicinities of both outfalls, but especially the Southern Works outfall. As discussed in Chapter 5, the communities are characterised by an abundance of capitellid polychaetes that are known to be tolerant of excessive organic matter accumulation in sediment (e.g. Méndez et al. 1997, Gómez Gesteira and Dauvin 2005). Further evidence comes from the observation that sediment near the Southern Works outfall and to a lesser degree the Central Works outfall is routinely discoloured (black) and tainted with the aroma of hydrogen sulphide (which imparts a rotten egg smell). Many samples also had a 'sewage smell' when retrieved (recorded in field data sheets that are available on request). Hydrogen sulphide is a product of the decomposition of organic matter under anoxic conditions and is highly toxic to most aquatic organisms at elevated concentrations. Anoxia develops in sediment when the rate of organic matter accumulation exceeds the rate at which this material can be processed by benthic biota. As a result, the rate of decomposition by aerobic bacteria increases, leading to a demand for oxygen that exceeds oxygen replenishment. It is these bacteria that produce the hydrogen sulphide. Some of the hydrogen sulphide reacts with divalent metals (e.g. iron, cadmium, copper, lead, nickel and zinc) to produce metal-sulphide complexes, which are highly insoluble under anoxic conditions. These metal-sulphides (but especially iron-sulphide) form a black or brown deposit, leading to the discoloration of sediment. In the 2011 survey, all sediment samples collected from the grid of stations spanning the Southern Works outfall were to varying degrees discoloured. Fifteen of the 30 sediment samples also had a strong aroma of hydrogen sulphide. In contrast, only five of the 30 sediment samples collected from the grid of stations spanning the Central Works outfall were discoloured, some of which had a faint aroma of hydrogen sulphide. This said, several other sediment samples smelled of sewage.

4.3.4. Chemical Oxygen Demand

Chemical oxygen demand is a measure of the oxygen equivalent of organic and inorganic materials that are susceptible to breakdown (oxidation) by a strong chemical oxidant. Chemical oxygen demand is generally higher than biological oxygen demand (oxygen demand by aerobic microorganisms) because chemical oxidation breaks down more compounds than microorganisms can. Biological oxygen demand is primarily from decaying animal and vegetable matter, while chemical oxygen demand results from oxidation of these materials in addition to compounds such as solvents, hydrocarbons, pesticides and soaps.

Long-term data generated by the CSIR through the monitoring of coastal waters in KwaZulu-Natal has shown that the chemical oxygen demand of sediment from anthropogenically un-impacted locations is usually positively correlated to the mud fraction of the sediment. This presumably occurs due to the accumulation of organic matter in sediment in a proportional manner to the mud fraction due to the similar winnowing or deposition of these fine-grained materials. As was the situation for the total organic content of sediment, this relationship can be modelled through linear regression analysis and the regression and its associated prediction limits can then be used to interpret whether sediment collected at (potentially) anthropogenically impacted areas has a higher than expected oxygen demand.

The relationship between the chemical oxygen demand and very fine-grained sand and mud fraction of sediment from the reference sites was strong (Figure 4.13). This allowed for the use of regression models and associated prediction limits for the interpretation of data in the same manner described above for total organic content. There was little difference in the number of sediment samples identified as having a higher than expected chemical oxygen demand based on relationships using either very fine-grained sand or mud as the normaliser. The majority of sediment samples collected from the grid of stations spanning the Southern Works outfall had a chemical oxygen demand that exceeded the maximum predicted from the relationship for the reference sites. Far fewer samples collected from the grid of stations spanning the Central Works outfall were identified as having a higher than expected chemical oxygen demand (Figure 4.13).

Average Enrichment Factors for chemical oxygen demand in sediment at the Central Works outfall sites were slightly higher compared to reference sites.

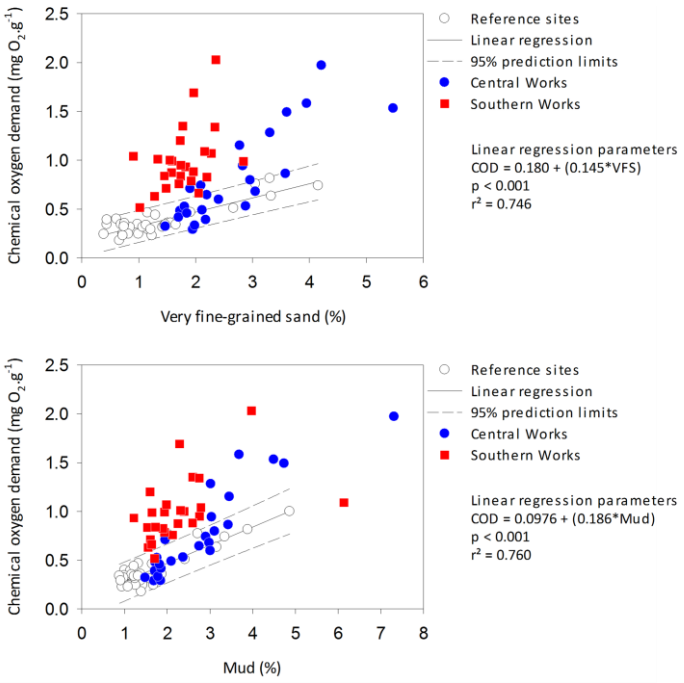

Figure 4.13. Relationship between chemical oxygen demand and very fine-grained sand mud fractions of sediment collected for the 2011 survey of the Durban outfalls monitoring programme.

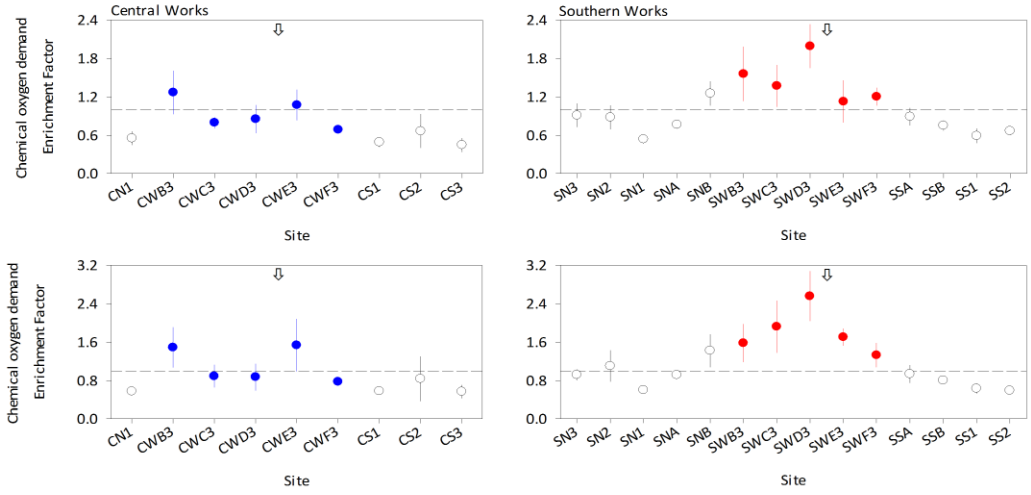

Figure 4.14. Average (± one standard deviation) chemical oxygen demand Enrichment Factors determined from regressions using very fine-grained sand (top) and mud (bottom) as the normaliser. Enrichment Factors >1 (horizontal dashed lines) indicate that the total organic content was higher than the expected range predicted for sediment from reference sites. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

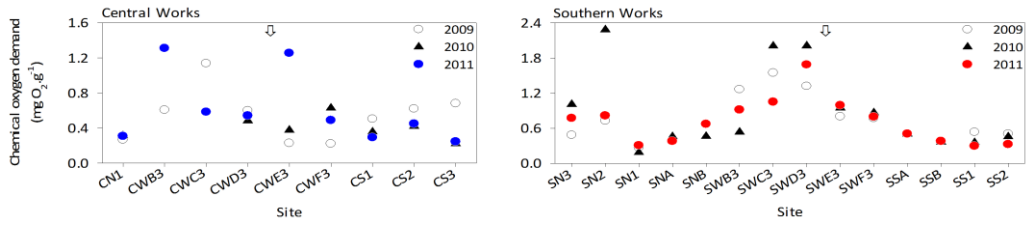

Figure 4.15. Comparison of the average chemical oxygen demand of sediment for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. Error bars are not included for presentation purposes. Arrows denote the positions of sites relative to the outfall diffusers.

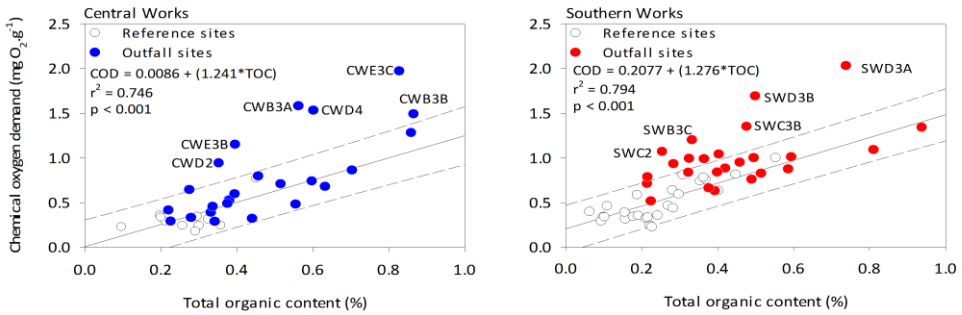

Figure 4.16. Relationship between the total organic content and chemical oxygen demand of sediment collected for the 2011 survey of the Durban outfalls monitoring programme.

The highest average Enrichment Factors were for sites CWB3 and CWE3, situated about 500 m to the north-northeast and 160 m to the south-southwest of the diffuser respectively (Figure 4.14). There was a very clear trend across reference and outfall sites for the Southern Works outfall. The highest average Enrichment Factor was for site SWD3, situated about 100 m to the north-northeast of the diffuser. Enrichment Factors then more or less progressively decreased with distance to the north-northeast and south-southwest of the outfall, the notable exception again being an increase at reference sites SN2 and SN3 (Figure 4.14). All average Enrichment Factors exceed a value of one for outfall sites and also for reference site SNB.

The trend for chemical oxygen demand in sediment at Central Works outfall and reference sites was broadly comparable to trends for the 2009 and 2010 surveys of the Durban outfalls monitoring programme, but very similar for the Southern Works outfall (Figure 4.15). The only exception was the anomalously high average Enrichment Factor for chemical oxygen demand measured at reference site SN2 in the 2010 survey.

There were strong and statistically highly significant linear relationships between the total organic content and chemical oxygen demand of sediment for reference sites for both outfalls (Figure 4.16). The relationships also allow for the identification of sediment samples with a chemical oxygen demand that was anomalously high relative to the total organic content of the sediment. The strong linear relationships for the reference sites provide strong evidence that the chemical oxygen demand at these sites was largely driven by the degradation of particulate organic material. However, the absence of a similarly strong linear relationship for sediment at several stations on the grids of stations spanning the diffuser sections of the outfalls suggests that, at these stations, unknown factors are also contributing to the oxygen demand. There was no pronounced spatial trend for samples that were identified as a having an anomalous chemical oxygen demand and this makes it difficult to decide on the cause of the elevated oxygen demand. There seems little doubt, however, that the higher chemical oxygen demand of sediment in the vicinities of the outfalls compared to reference sites is attributable to effluent derived material that is accumulating on the seabed.

4.3.5. Metals

Determining whether sediment is contaminated by some chemicals is easy since these only have an anthropogenic origin (e.g. polychlorinated biphenyls). In other words, the mere presence of these chemicals is indicative of contamination. Determining whether sediment is metal contaminated is far more complicated. Perhaps the most important reason is that metals are a ubiquitous, naturally occurring component of sediment. The presence of metals in sediment does not, therefore, provide an indication that the sediment is metal contaminated. Determining whether sediment is metal contaminated is further complicated by the fact that metal concentrations in uncontaminated sediment can vary by orders of magnitude over relatively small spatial scales depending on sediment mineralogy, granulometry and organic content amongst other factors (Loring and Rantala 1992, Kersten and Smedes 2002). High metal concentrations in sediment do not automatically imply that the sediment is metal contaminated, but may simply reflect the natural mineralogical composition of the parent material

and granulometry and organic content of the host sediment. As a still further complication, despite input and transport dissimilarities naturally occurring and anthropogenically introduced, metals tend to accumulate in the same areas (Hanson et al. 1993).

To meaningfully interpret metal concentrations in sediment, the mineralogic and granulometric factors that influence the natural variation of metal concentrations in sediment must be compensated for before naturally occurring concentrations can be differentiated from anthropogenically introduced concentrations (Kersten and Smedes 2002). This can be accomplished through the procedure of normalisation, which mathematically normalises metal concentrations to a co-occurring conservative element (the normaliser, sometimes referred to as reference element) that provides a tracer of crustal decomposition (Kersten and Smedes 2002).

The basis for geochemical normalisation is that while absolute metal concentrations vary between crustal material from one region to another, the relative proportions of metals within crustal material from a particular region tend to be fairly constant (e.g. Turekian and Wedepohl 1961, Taylor and McLennan 1981, Martin and Whitfield 1983, Wedepohl 1995, Kersten and Smedes 2002). Since there is relatively little fractionation between metals and aluminosilicates during weathering (Schropp and Windom 1988), metal concentrations in sediment tend to reflect the relative proportions of metals in the parent material. This permits the modelling of relationships between metal concentrations and co-occurring normaliser concentrations through simple or multiple linear regression analysis. By quantifying the variability in metal concentrations around the regression line, through the definition of prediction limits, the range in variability of baseline metal concentrations for an area can be defined. Similarly normalised metal concentrations measured in sediment collected from areas where metal enrichment is suspected can then be compared to the models, to interpret whether or not the sediment is metal enriched (i.e. to identify 'excess' metal concentrations).

The use of a metal as a proxy for the natural metal-bearing phases of sediment (i.e. aluminosilicates) requires that the metal meet several assumptions, namely that it: 1. is highly refractory, 2. is structurally combined to one or more of the major metal-bearing phases of sediment, 3. co-varies in proportion to the naturally occurring concentrations of metals of interest, 4. is insensitive to inputs from anthropogenic sources, and 5. is stable and not subject to environmental influences such as reduction/oxidation, adsorption/desorption and other diagenetic processes that may alter sediment concentrations (Luoma 1990). Several metals have been used as normalisers, including aluminium (see for example Schropp et al. 1990, Hanson et al. 1993, Daskalakis and O'Connor 1995, Cooke and Drury 1998, Weisberg et al. 2000, Roach 2005, Newman and Watling 2007, Newman et al. manuscript submitted), iron (Daskalakis and O'Connor 1995, Schiff and Weisberg 1999, Tanner et al. 2000, Cobelo-García and Prego 2003, Newman and Watling 2007, Newman et al. manuscript submitted), lithium (Loring 1990, 1991, Aloupi and Angelidis 2001, Veinott et al. 2001), rubidium (Grant and Middleton 1990), caesium (Ackermann 1980, Roussiez et al. 2005) and cobalt (Matthai and Birch 2001, Matthai et al. 2002, Newman et al. manuscript submitted) amongst others. Aluminium and iron are, however, most frequently used as normalisers. Of these, aluminium is considered the better normaliser since it is a major constituent of fine-grained aluminosilicates (silts and clays), with which the bulk of trace metals are associated. Sand, in contrast, is comprised predominantly of metal poor quartz (silica). Aluminium concentrations are, therefore, usually strongly inversely correlated to sediment grain size and are strongly positively correlated to co-occurring metal concentrations. Aluminium is also stable and not affected by early diagenic processes and strong redox effects commonly observed in sediments (Kersten and Smedes 2000), and is highly refractory. Although iron is not as tightly incorporated into the crystal lattice of aluminosilicates as aluminium, iron oxide coatings, which serve as a host for metals, are usually associated with sediments in definite quantities related to the sediment surface area. The concentration of iron consequently usually also exhibits a strong inverse correlation to grain size and positive correlation to co-occurring metal concentrations (Daskalakis and O'Connor 1995, Kersten and Smedes 2002). A potential limitation for the use of iron is that it may be highly mobile in anoxic sediments, leading to its enrichment at the

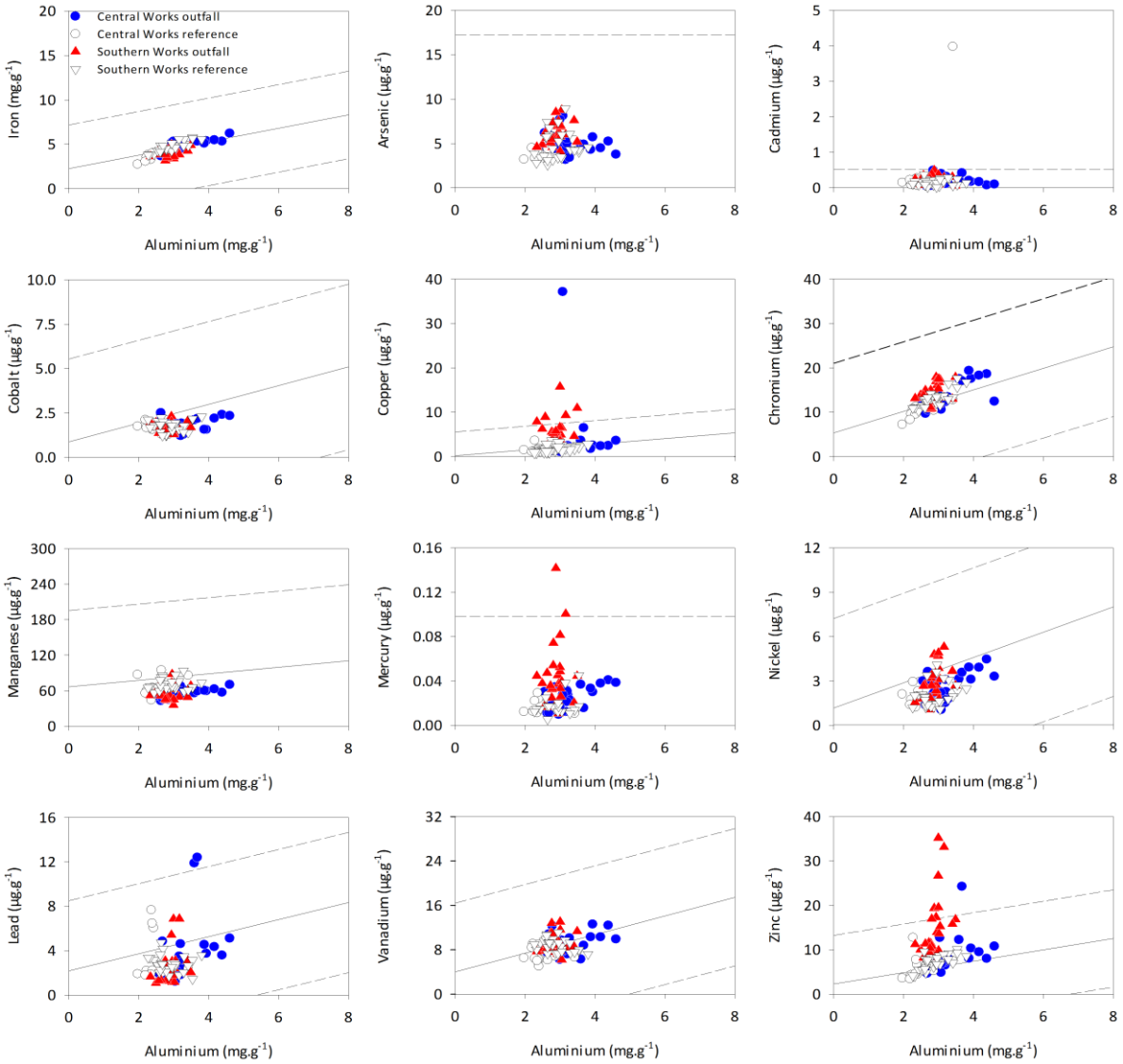

Figure 4.17. Baseline metal concentration models and baseline metal concentrations for sediment from the KwaZulu-Natal coastal waters overlaid with metal concentrations in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Horizontal dashed lines in arsenic, cadmium and mercury plots indicate the baseline concentration above which enrichment of sediment by these metals can be inferred.

sediment surface through the deposition of iron oxides (Finney and Huh 1989) or in deeper sediments through co-precipitation with sulphides (Gobeil et al. 1997). This may lead to an underestimation of enrichment of other metals not similarly affected when iron is used as a normaliser.

The natural concentrations of aluminium and iron, respectively the third and fourth most abundant elements in the earth's crust (Wedepohl 1995), are orders of magnitude higher in sediment than concentrations of the trace metals typically of concern from a toxicological perspective ($mg.g^{-1}$ versus $\mu g.g^{-1}$ concentrations respectively). These high natural concentrations 'swamp' the usually low inputs of these metals to the environment from anthropogenic sources, and their concentrations are therefore likely to remain relatively unchanged even in anthropogenically impacted areas. The naturally low concentration trace metals are, in contrast, far more sensitive to anthropogenic inputs, with the result that the ratio between the metal of concern and the normaliser is altered. Normalisation serves to detect such alteration in ratios, by comparing metal concentration ratios measured in sediment from minimally- or un-contaminated locations to those in potentially metal enriched sediment.

As stated above, aluminium is the most commonly used normaliser for the definition of baseline metal concentration models. The CSIR has developed aluminium, iron, cobalt and vanadium normalised

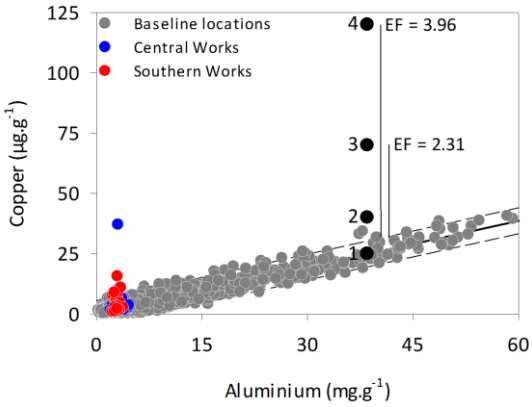

Figure 4.18. Baseline model for copper in sediment from the KwaZulu-Natal coast. The grey symbols represent concentrations used to define the baseline model, while red and blue symbols represent concentrations measured in sediment collected at Central Works and Southern Works outfall impact and reference sites in the 2011 survey of the Durban outfalls monitoring programme. Black symbols represent four hypothetical scenarios: 1. concentration falls within the model upper and lower 99% prediction limits (stippled lines) and is therefore not enriched; 2, 3 and 4. concentrations exceed model upper 99% prediction limit and reflect various levels of enrichment that can broadly be defined from low (2) through to high (4). Situations 3 and 4 would be interpreted as reflecting enrichment through contamination with a high level of confidence. Enrichment Factors for two of the scenarios are indicated.

baseline metal concentration models for sediment along the KwaZulu-Natal coast (Newman et al., manuscript submitted). The aluminium normalised baseline models are typically used by the CSIR to interpret metal concentrations measured in test sediment samples, while the iron, cobalt and vanadium normalised models are used to verify that aluminium itself is not enriched in sediment. The natural concentrations of cobalt and vanadium in sediment are far lower than those of aluminium and iron, providing a far greater potential for significant concentration alteration through anthropogenic contributions and, hence, negating their use as normalisers. Cobalt and vanadium have limited industrial application in South Africa, however, and are typically not contaminants of sediment from local coastal waters, making them suitable validation elements.

Figure 4.17 presents aluminium normalised baseline concentration models for iron, cobalt, copper, chromium, manganese, nickel, lead, vanadium and zinc in sediment from KwaZulu-Natal coastal waters overlaid with aluminium normalised metal concentrations in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Concentrations of arsenic[7], cadmium and mercury in sediment from many coastal locations in KwaZulu-Natal are weakly to very weakly correlated to co-occurring aluminium and iron concentrations. This suggests that a significant proportion of the concentrations of these metals has a non-(inorganic)detrital origin. Normalisation could not, therefore, be used to define baseline models for these metals. The weak relationships between the latter metals and aluminium and iron concentrations is also evident for other regions of the South African coast(see Newman and Watling 2007), and indeed for sediment from many regions of the world. Baseline concentrations above which enrichment of sediment by arsenic, cadmium and mercury can be inferred were therefore defined using cumulative probability and univariate distribution plots of concentrations, at 17.21 µg.g^{-1} for arsenic, 0.518 µg.g^{-1} for cadmium and 0.098 µg.g^{-1} for mercury (Figure 4.17). The use of cumulative probability and univariate distribution plots for defining baseline metal concentrations is a more subjective procedure compared to the definition of baseline metal concentration models through normalisation. Hence, there is somewhat less confidence regarding the reliability of baseline concentrations for arsenic, cadmium and mercury.

The baseline models comprise a regression line and upper and lower 99% prediction limits (oblique solid and dashed lines respectively in Figure 4.17). The regression line defines the average concentration for a metal at co-occurring aluminium concentrations in sediment from baseline (uncontaminated) locations in KwaZulu-Natal coastal waters, while the upper and lower prediction limits define the range around this average concentration within which 99% of the concentrations should theoretically fall if the sediment is uncontaminated. Concentrations that plot above the upper prediction limit represent enrichment (see hypothetical scenarios provided in Figure 4.18). The reader will note that concentrations measured in sediment collected for the 2011 survey of the Durban outfalls monitoring programme plot to the extreme left of the x-axis in

[7] The reader should note that while arsenic is technically a metalloid (i.e. a semi-metal), for the sake of simplicity it is referred to as a metal in this report.

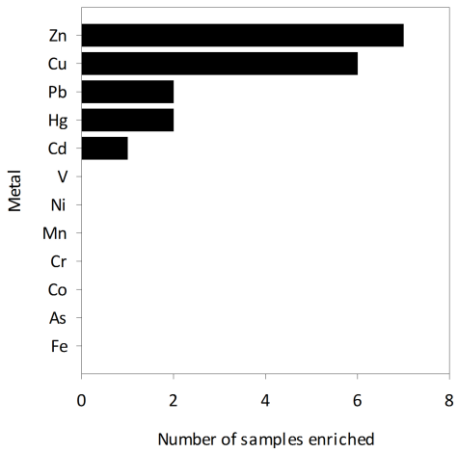

Figure 4.19. Number of sediment samples enriched with different metals in the 2011 survey of the Durban outfalls monitoring programme.

Figure 4.18. This reflects the fact that the sediment was dominated by metal deficient sand. Thus, the plots presented in Figure 4.17 incorporate only a small portion of the baseline models.

It is important to note that a metal concentration that plots above a model upper prediction limit does not necessarily imply that the enriched concentration is enhanced through anthropogenic inputs (i.e. reflects contamination), but rather that the concentration is atypical of the data set used to define the model. Several possible reasons in addition to anthropogenic inputs may lead to a metal concentration exceeding a model upper prediction limit. These include analytical errors, poor model assumptions, the probability that metal concentrations in some samples will naturally exceed the upper prediction limit (in a normally distributed population, at the 99% prediction limit 1 in every 100 concentrations could conceivably naturally exceed the limit), and natural enrichment not captured by the baseline data set (Schropp et al. 1990, Rae and Allen 1993).

Interpretation of enrichment thus requires consideration of ancillary factors, including possible biogeochemical processes leading to natural enrichment, the absolute difference between a measured metal concentration and the model upper prediction limit, the location of enriched sediment relative to known or potential anthropogenic metal sources, and assessment of the number of metals at a site that exceed model upper prediction limits. The larger the discrepancy between a measured metal concentration and the model upper prediction limit (see Figure 4.18) and the greater the number of metals enriched at a particular station the higher the likelihood that the metal concentration is enriched through anthropogenic inputs (i.e. reflects contamination). Typically, in situations of contamination, several metals are enriched in sediment from a station rather than just a single metal, particularly where the metal source is diffuse (e.g. stormwater runoff).

All but 19 (or 1.64%) of the 1157 metal concentrations measured in sediment collected for the 2011 survey of the Durban outfalls monitoring programme fall within baseline model upper and lower prediction limits or below baseline concentrations (Figures 4.17). Five of the 12 metals analysed were present at enriched concentrations, with copper and zinc most frequently enriched (Figure 4.19).

Although few metal concentrations exceed baseline model upper prediction limits or baseline concentrations and are thus interpreted as enriched, there is evidence that the concentrations of several metals in sediment in the immediate vicinities of the outfalls are higher compared to the reference sites. For example, copper concentrations in sediment at numerous stations on the grid of stations spanning the diffuser section of the Southern Works outfall, while not exceeding the model upper prediction limit, were nevertheless higher compared to concentrations at other stations and at reference sites (see Figure 4.17). It is important to note that the baseline models and baseline concentrations were established from metal concentration data collected over a large area of the KwaZulu-Natal coast. Thus, small, naturally occurring differences in metal concentrations between sites are incorporated into the models. Consequently, the slope of a regression model and the width of prediction limits at an area specific level might differ slightly to the regional model. Baseline models established specifically for the Bluff area of Durban might, therefore, have narrower prediction limits and hence interpret the slightly higher copper concentrations mentioned above as enriched. Unfortunately, there are too few data to define area specific baseline models and baseline concentrations for most areas of the Kwazulu-Natal coast. There is, however, an alternate approach that can be used to evaluate

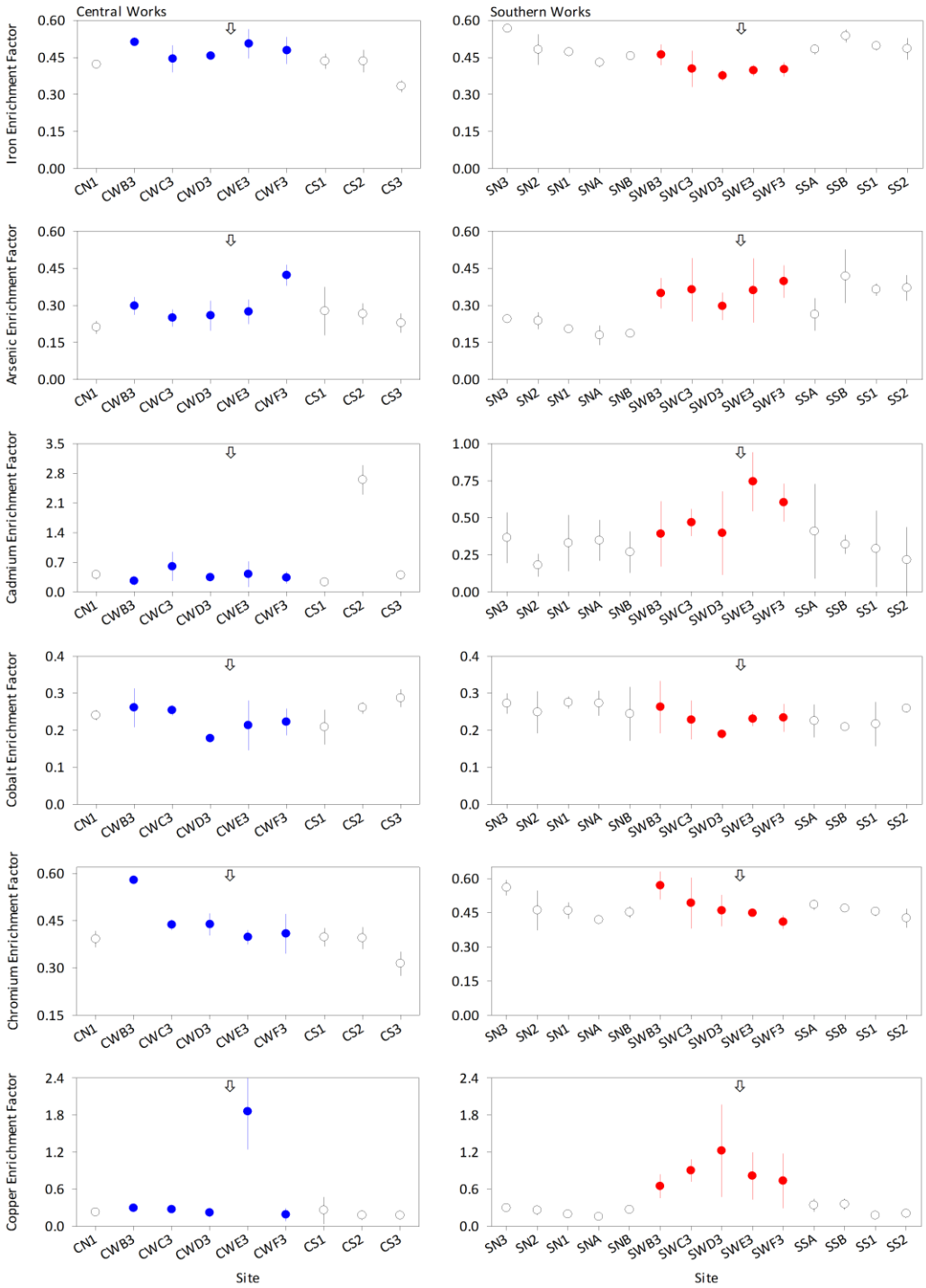

Figure 4.20. Average Enrichment Factors (± one standard deviation) for metals in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

whether the slightly higher metal concentrations are indicative of an anthropogenic impact, namely through the calculation of Enrichment Factors.

Enrichment Factors are typically used to contextualise the magnitude of metal concentration exceedance of a baseline model upper prediction limit. However, they can just as easily be used to examine for differences between data that do not exceed the prediction limit. The Enrichment Factor 'corrects' for natural differences in metal concentrations in sediment brought about by differences in its granulometry. In this manner, anomalous metal concentrations can be identified through the application of various statistical procedures. Enrichment Factors (EF) were calculated

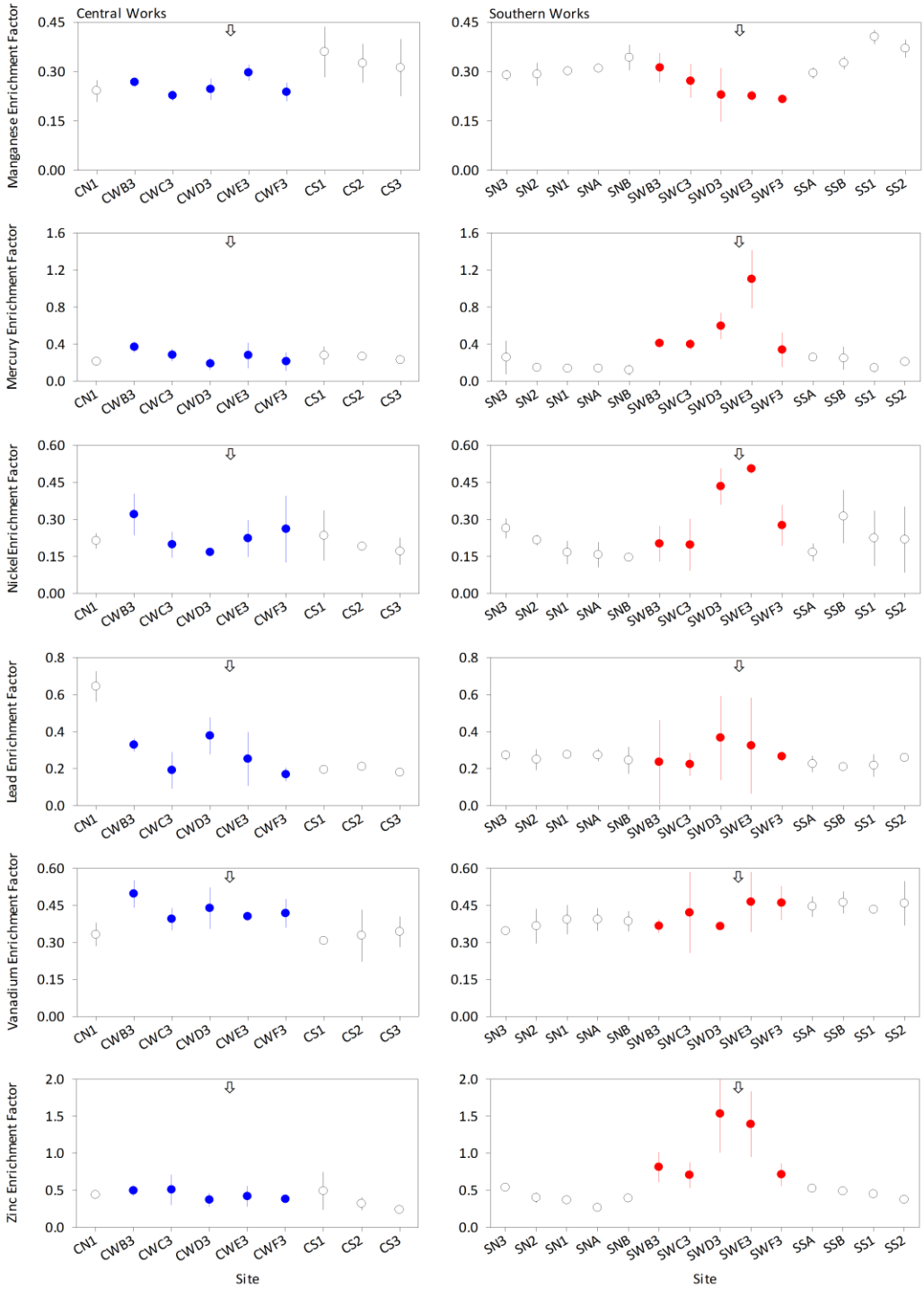

Figure 4.20 continued. Average Enrichment Factors (± one standard deviation) for metals in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

as

$$EF = [M]_{obs}/[M]_{pred}$$

where $[M]_{obs}$ is the metal concentration in the sediment sample in question and $[M]_{pred}$ is the metal concentration predicted at the baseline model upper prediction limit at the corresponding aluminium concentration measured in the sediment sample. The Enrichment Factor is a unitless value that indicates by how many times a metal concentration exceeds (or is lower than) the concentration predicted at the baseline model upper prediction limit in granulometrically equivalent sediment. An Enrichment Factor ≤1 denotes that there is no enrichment while an Enrichment Factor >1 denotes

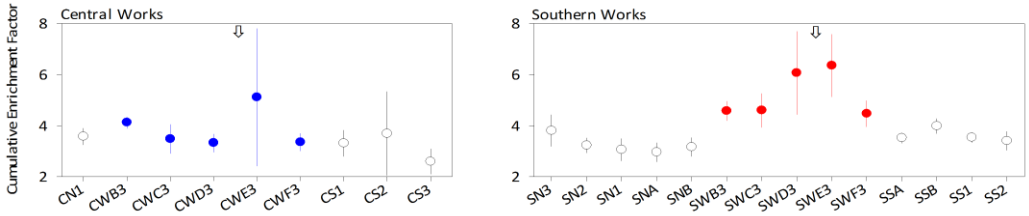

Figure 4.21. Average cumulative Enrichment Factors (± one standard deviation) for metals in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

that the metal is at a concentration higher than expected for uncontaminated sediment from the region in question. An Enrichment Factor = 1.50, for example, indicates that the metal concentration is 1.5, or 50%, higher than the predicted concentration at the model upper prediction limit.

For those metals where baseline models could not be defined (i.e. arsenic, cadmium and mercury), the Enrichment Factor was calculated as

$$EF = [M]_{obs}/[M]_{base}$$

where $[M]_{obs}$ is the metal concentration in the sediment sample and $[M]_{base}$ is the baseline concentration above which enrichment of sediment from KwaZulu-Natal coastal waters by these metals can be inferred.

Considering that the concentrations of most metals in sediment collected at the Central Works and Southern Works outfall sites and associated reference sites for the 2011 survey fall within baseline model prediction limits or below baseline concentrations, Enrichment Factors for most metals in most samples were not surprisingly ≤1. The highest Enrichment Factor was for a cadmium concentration in sediment collected at reference site CS2, at 7.67. In other words, this concentration was 7.67 times higher than the baseline concentration. The source of cadmium at this site is uncertain, but is interesting since no other metals were present in sediment at anomalous concentrations at this site, and cadmium concentrations in the two other replicate sediment samples collected at this site were not anomalous.

As stated previously, the Enrichment Factors need not only be used to quantify the magnitude of baseline model prediction limit exceedance by a metal concentration, but can also be used to evaluate other anomalies in data. Figure 4.20 presents average Enrichment Factors for different metals at the reference sites and at sites in the immediate vicinities of the Central Works and Southern Works outfalls. As is evident, the average Enrichment Factors for several metals (cadmium, copper, mercury, nickel and zinc) were higher at the Central Works and/or Southern Works outfall sites compared to the reference sites. The most pronounced differences were for the Southern Works outfall, especially sites SWD3 and SWE3. These sites are situated 100 m to the north-northeast and south-southwest of the diffuser respectively.

Cumulative Enrichment Factors, which were calculated by summing Enrichment Factors for different metals in a sample and then expressing these as an average Enrichment Factor, provide further evidence for metal contamination of sediment near the outfalls, but again especially near the Southern Works outfall (Figure 4.21). Sites near the outfalls were also most frequently enriched by different metals (Figure 4.22).

Average Enrichment Factors for reference sites SN3 and SN2 were often somewhat higher than Enrichment Factors for other Southern Works reference sites and in some cases were of a similar order as for outfall sites (Figure 4.21). This same trend was evident in previous surveys (see below). It is uncertain whether this reflects the influence of the higher mud content at these sites or whether effluent particulate material and adsorbed metals are settling from the water column in this area. Discussions above for the mud fraction, total organic content and chemical oxygen demand of sediment have alluded to anomalies at these sites and it is

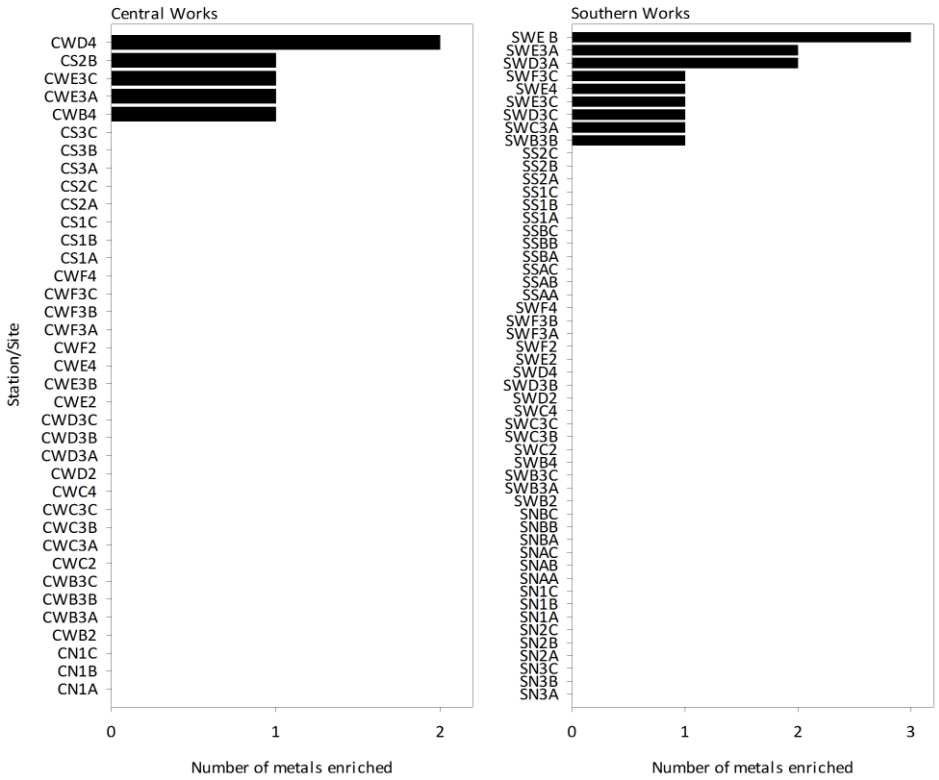

Figure 4.22. Number of metals enriched in sediment at different stations/sites for the 2011 survey of the Durban outfalls monitoring programme.

apparent that this also extends to metals.

Concentrations of all but two metals were well below relevant Level 1 sediment quality guidelines for the protection of aquatic life (see Table 4.1). Level 1 sediment quality guidelines are used to indicate levels of contamination where adverse effects to sediment-dwelling organisms can first be anticipated, although these will probably not be significant adverse impacts. The exceptions mentioned above were a cadmium concentration in one of the three replicate sediment samples collected at reference site CS3, which substantially exceed the DEA Special Care Level and Long et al. (1995) Effects Range Low guidelines, and a copper concentration in one of the three replicate sediment samples collected at site CWE3, which marginally exceeds the Long et al. (1995) Effects Range Low guideline. No metal concentrations exceed Level 2 guidelines. Theoretically, then, there is little probability that metal concentrations in sediment were exerting direct toxic effects to sediment-dwelling organisms.

Trends in average Enrichment Factors for most metals and cumulative Enrichment Factors in the 2011 survey were comparable to trends for the 2009 and 2010 surveys of the Durban outfalls monitoring programme (Figure 4.23 and 4.24). In other words, there was generally little change in metal contamination of sediment in the study area between 2009 and 2011. Although the data are not presented, the trend in metal contamination of sediment in the immediate vicinities of the outfalls has remained broadly comparable since the early 1990's in terms of the actual metals enriched and Enrichment Factors.

4.3.6. Organic Chemicals

A wide suite of organic chemicals were analysed in sediment samples (see Table 1.4). Several of the chemicals are on the Stockholm Convention list of banned substances, because of the known ecological and human health risks they pose. Of the various organic chemicals analysed, only polycyclic aromatic hydrocarbons were detected at concentrations exceeding the method detection limit and then only at four sites (Figure 4.25). Three of the sites (CWF3, SWD3, SWE3) were in the immediate vicinities of the outfalls. This suggests an effluent source. The fourth site was reference site CN1, situated about 2000 m to the north-northeast of the Central Works outfall.

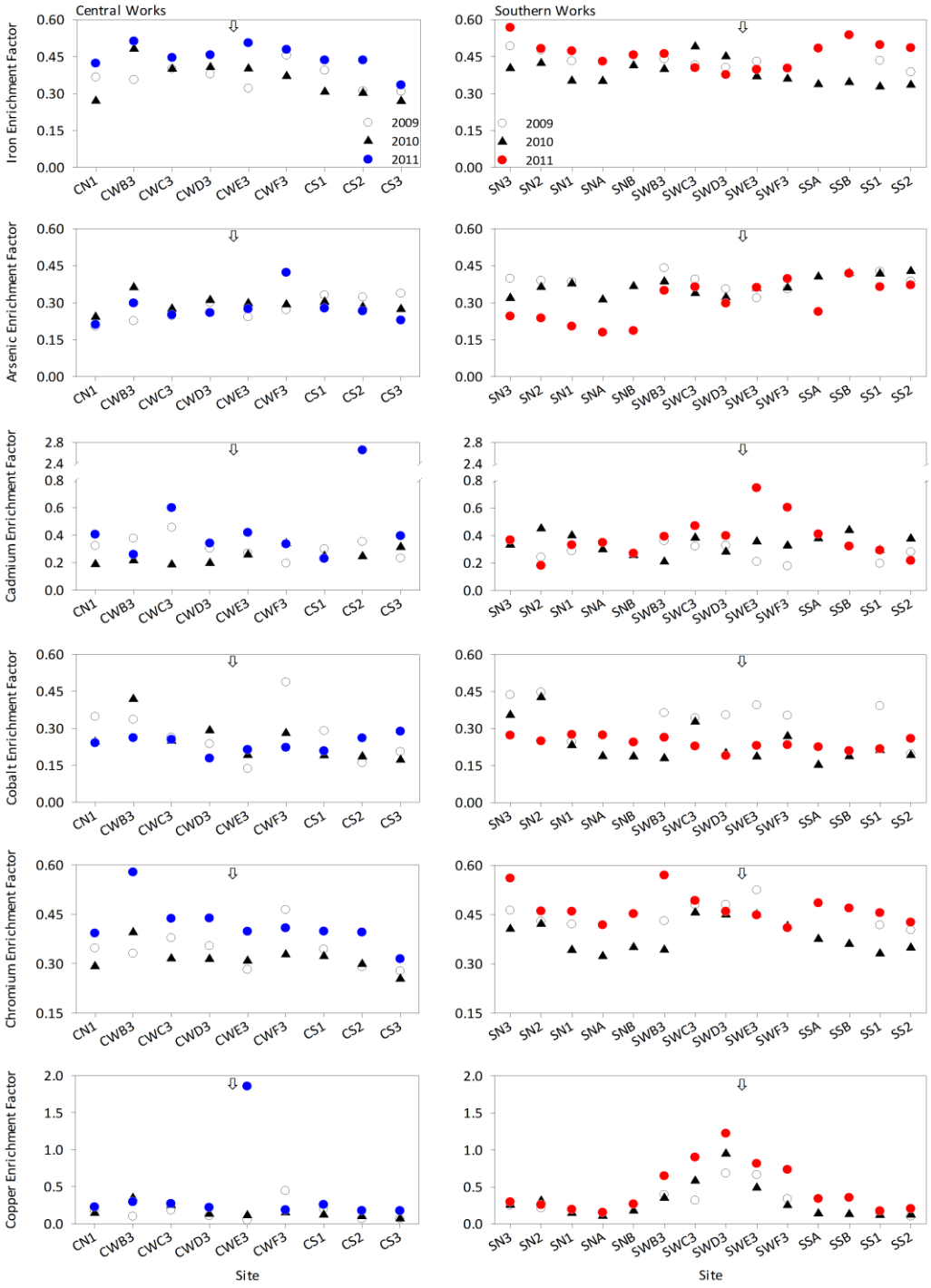

Figure 4.23. Average Enrichment Factors for metals in sediment collected for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. Error bars are not included for presentation purposes. Arrows denote the positions of sites relative to the outfall diffusers.

The source of the hydrocarbons at this site is uncertain, especially considering that no other sediment variables were identified as anomalous at this site. Effluent cannot, however, be ruled out as a source, even though this seems remote. Site CN1 is situated near the dredge spoil disposal ground, where sediment dredged from the Port of Durban is disposed. The disposal of dredged spoil may be the source of the hydrocarbons.

Low molecular weight isomers were infrequently detected, which probably reflects the higher solubility of these isomers compared to high molecular weight isomers, which are more prone to accumulation in sediment. Also, low molecular weight isomers are more readily degraded by microorganisms. The concentrations of all polycyclic aromatic hydrocarbon isomer and total isomer concentrations were below the Effects Range Low

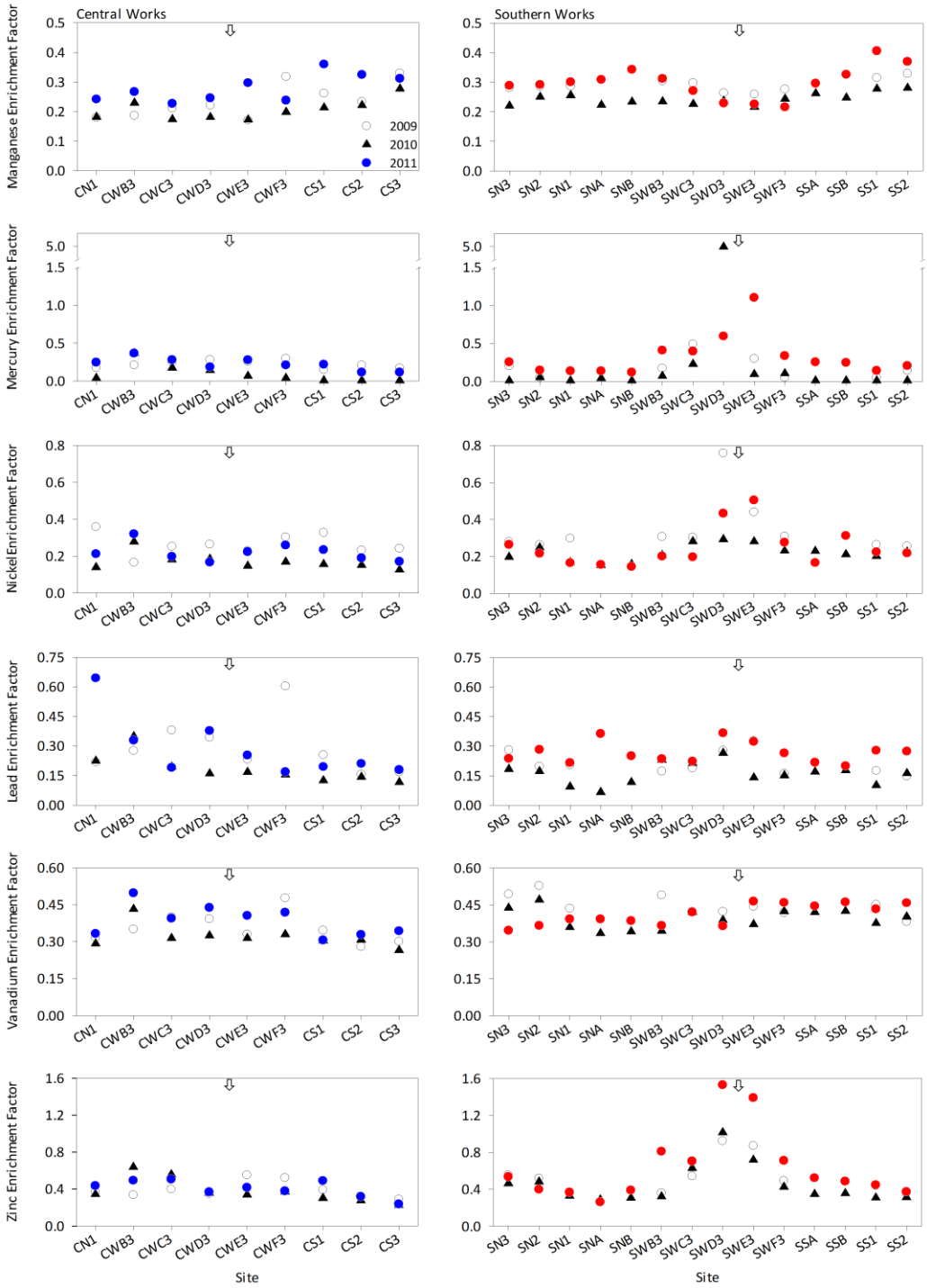

Figure 4.23 continued. Average Enrichment Factors for metals in sediment collected for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. Error bars are not included for presentation purposes. Arrows denote the positions of sites relative to the outfall diffusers.

guideline of the sediment quality guidelines defined by Long et al. (1995; see Table 4.1) with the exception of pyrene at site SWD3, which marginally exceeds the guideline. Theoretically, then, the polycyclic aromatic hydrocarbon concentrations pose little toxic risk to sediment-dwelling organisms. The trend for organic chemicals in sediment for the 2011 survey is similar to the trend for the 2009 and 2010 surveys, when polycyclic aromatic hydrocarbons were detected in only a few samples. Interestingly, polycyclic aromatic hydrocarbons were also detected at reference site CN1 in the 2010 survey (CSIR 2010, CSIR 2011).

4.3.7. Porewater Toxicity Testing

Porewater at all sites except CN1 and CWD3 was toxic to sea urchin gametes (Figure 4.26). The degree of toxicity for Central Works sites was, however,

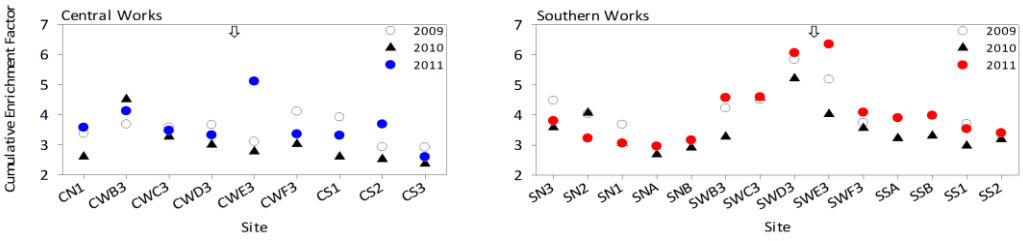

Figure 4.24. Average cumulative Enrichment Factors for metals in sediment collected for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. Error bars are not included for presentation purposes. Arrows denote the positions of sites relative to the outfall diffusers.

usually minimal, with fertilisation inhibition typically in the order of 2% but up to 8.2%. The exception was reference site CS3, where on average less than 10% of gametes were fertilised (compared to 95% for the control treatment). The trend in toxicity was broadly comparable to that for the 2010 survey of the Durban outfalls monitoring programme, although no toxicity was evident at reference site CS3 in the latter survey (Figure 4.27).

Porewater toxicity at all reference sites for the Southern Works outfall was of a broadly comparable order and showed no particular spatial trend (Figure 4.26). No gametes fertilised following exposure to porewater for outfall sites. In the 2010 survey, porewater from only two of the outfall sites was toxic (Figure 4.27). In other words, the toxicity evident in the 2011 survey represents an increase in spatial extent compared to the 2010 survey.

Although there was a fairly strong relationship between total ammonia-N concentrations in porewater and fertilisation inhibition for many samples, as illustrated by the generally linear relationship in Figure 4.28, this was not the case for most of the Southern Works outfall sites and for the Central Works outfall reference site mentioned above. In other words, while total ammonia-N in porewater appears to account for toxicity at some sites it does not explain the toxicity at all sites. Other parameters, such as the presence of sulphide, are probably also contributing to the toxicity. Hydrogen sulphide is extremely toxic to most aquatic organisms at elevated concentrations and was clearly present in sediment as noted by fieldwork staff who detected its aroma in many samples collected near the outfalls. It is important to note that ammonia (both in the form NH_3-N and NH_4^+-N) occurs naturally in sediment and its presence does not necessarily indicate an effluent discharge impact. Nevertheless, some high concentrations near the outfalls almost certainly do reflect such an impact. Future surveys will need to measure for the presence of sulphide to provide a better understanding of the cause of the toxicity.

4.4. Conclusions

Considering the physical, chemical and biological indicators of environmental condition in a weight of evidence approach provides clear evidence that the discharge of effluent from the Central Works and Southern Works outfalls was impacting on the benthic environment in the study area for the period between the 2010 and 2011 surveys of the Durban outfalls monitoring programme. The presence of faecal indicator bacteria in sediment at all stations/sites provides evidence that effluent was impinging on the benthic environment across the study area. However, faecal indicator bacteria were far more prevalent in sediment in the immediate vicinities of the outfalls, and more so for the Southern Works outfall compared to Central Works outfall. Particulate organic material is accumulating in sediment in the vicinities of the outfalls. This is driving an increased chemical oxygen demand, to the extent that anoxic conditions have developed in sediment at sites in the immediate vicinity of the Southern Works outfall. Metal concentrations in sediment in the immediate vicinities of the outfalls were elevated compared to reference sites. This said, metal concentrations were generally very low. Polycyclic aromatic hydrocarbons were more frequently measured at concentrations exceeding the method detection limit in the immediate vicinities of the outfalls, but only at four of the 23 sites sampled. The concentrations were, however, very low.

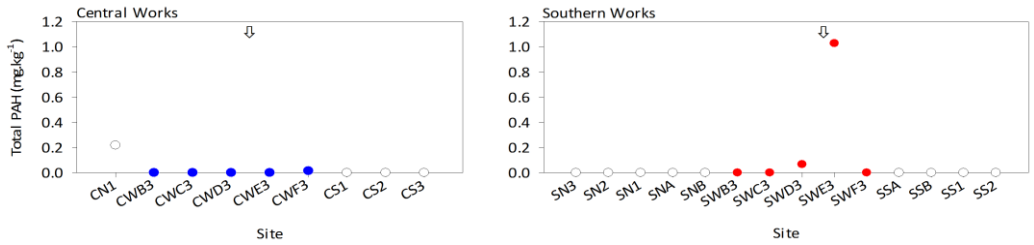

Figure 4.25. Total polycyclic aromatic hydrocarbon (PAH) concentrations in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

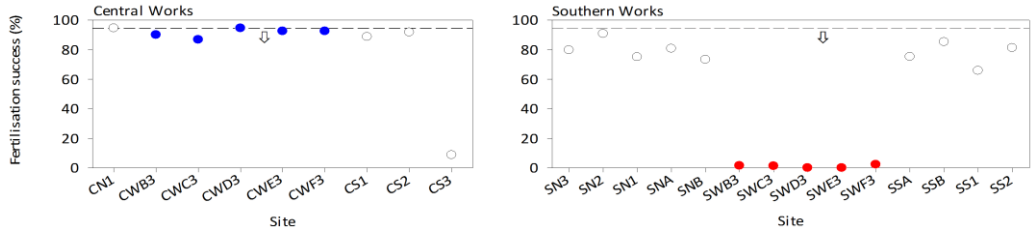

Figure 4.26. Fertilisation success of sea urchin gametes exposed to porewater in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers. The horizontal dashed lines denote the average fertilisation success for the control treatment. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

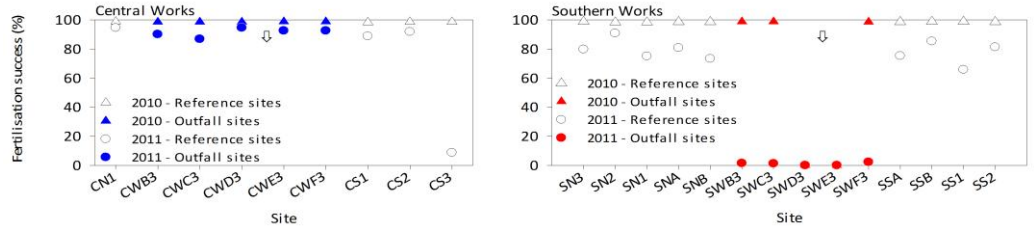

Figure 4.27. Fertilisation success of sea urchin gametes exposed to porewater in sediment collected for the 2010 and 2011 surveys of the Durban outfalls monitoring programme. Arrows denote the positions of sites relative to the outfall diffusers.

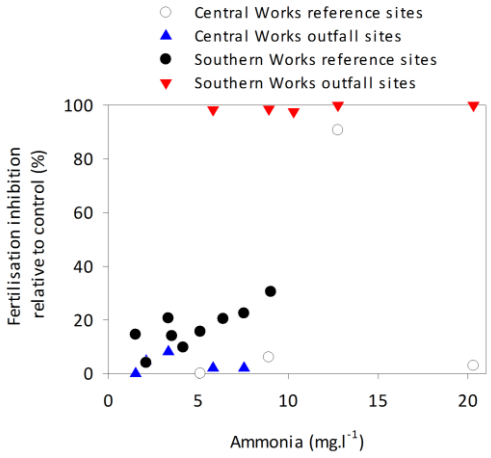

Figure 4.28. Relationship between the ammonia concentration of porewater in sediment collected for the 2011 survey of the Durban outfalls monitoring programme and the fertilisation inhibition of sea urchin gametes exposed to the porewater.

Although there is evidence for the contamination of sediment in the immediate vicinities of the outfalls by metals and polycyclic aromatic hydrocarbons, comparison of concentrations to sediment quality guidelines suggests there is theoretically little probability that these contaminants are adversely impacting benthic macrofaunal communities through direct toxicity.

Sediment porewater at the majority of outfall and reference sites was, however, toxic to sea urchin gametes. The magnitude of toxicity at Central Works outfall and reference sites was low with the exception of a reference site situated 6000 m to the southwest of the outfall. The toxicity of sediment porewater at Southern Works outfall reference sites was comparable and relatively mild, but was very high at sites situated in the immediate vicinity of the outfall. The cause of the toxicity could not be satisfactorily attributed to ammonia concentrations in porewater, nor to concentrations of metals and polycyclic aromatic hydrocarbons in sediment. There is a possibility that the high toxicity of porewater at some sites in the immediate vicinity of the Southern Works outfall may be due to the presence of hydrogen sulphide, which is highly toxic to most marine organisms.

There was little difference in trends for most of the indicators of benthic environmental condition between the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. In other words, the impact of effluent discharge from the Central Works and Southern Works outfalls on the physico-chemistry of the benthic environment in the study area has remained broadly comparable over the latter period.

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4.6. Appendices

Appendix 4.6.1. Grain size composition, total organic content and chemical oxygen demand of sediment collected for the 2011 survey of the Durban outfalls monitoring programme. VCS = very coarse-grained sand, CS = coarse-grained sand, MS = medium-grained sand, FS = fine-grained sand, VFS = very fine-grained sand, TOC = total organic content, COD = chemical oxygen demand.

Sample	Gravel (%)	VCS (%)	CS (%)	MS (%)	FS (%)	VFS (%)	Mud (%)	Mean size (mm)	TOC (%)	COD (mg O_2.g^{-1})
CN1A	0.19	0.66	4.74	54.98	36.54	1.51	1.38	0.28	0.20	0.358
CN1B	0.06	0.57	4.82	60.61	31.53	0.99	1.42	0.29	0.26	0.245
CN1C	0.14	0.78	6.72	55.66	33.90	1.42	1.37	0.29	0.33	0.314
CWB2	0.10	0.48	5.24	46.14	44.41	1.95	1.69	0.26	0.34	0.288
CWB3A	0.00	0.15	2.59	43.83	45.78	3.96	3.69	0.24	0.56	1.580
CWB3B	0.20	0.47	3.98	44.22	42.79	3.60	4.74	0.25	0.87	1.490
CWB3C	0.00	0.51	3.11	42.09	47.28	3.58	3.43	0.24	0.70	0.861
CWB4	0.14	0.92	2.65	39.13	50.85	3.31	3.02	0.24	0.86	1.280
CWC2	0.08	0.47	4.33	51.91	40.25	1.47	1.49	0.27	0.44	0.321
CWC3A	0.00	0.58	4.01	53.70	38.26	1.73	1.71	0.28	0.56	0.480
CWC3B	0.05	0.74	3.73	49.16	41.32	2.10	2.90	0.26	0.60	0.740
CWC3C	0.00	0.47	4.12	48.37	41.79	2.88	2.37	0.27	0.38	0.529
CWC4	0.05	0.57	2.64	48.19	42.52	3.05	2.98	0.25	0.63	0.679
CWD2	0.04	0.75	4.35	45.16	43.83	2.83	3.04	0.25	0.35	0.941
CWD3A	0.35	6.54	13.56	33.69	42.00	1.91	1.95	0.31	0.52	0.708
CWD3B	0.34	1.66	4.12	38.59	51.72	1.81	1.76	0.25	0.38	0.524
CWD3C	0.09	0.82	2.77	37.43	55.00	2.18	1.71	0.24	0.33	0.389
CWD4	0.29	0.77	2.19	34.74	52.04	5.47	4.49	0.22	0.60	1.530
CWE2	0.06	0.41	2.66	42.22	50.85	1.95	1.85	0.25	0.23	0.290
CWE3A	0.11	0.50	2.11	45.32	47.01	2.20	2.75	0.25	0.28	0.643
CWE3B	0.14	0.58	2.24	44.09	46.71	2.78	3.45	0.26	0.40	1.150
CWE3C	0.12	0.59	2.15	41.22	44.39	4.21	7.32	0.23	0.83	1.970
CWE4	0.06	0.58	2.60	47.24	43.45	2.96	3.11	0.26	0.46	0.795
CWF2	0.07	0.29	2.12	47.67	46.08	1.99	1.79	0.25	0.28	0.331
CWF3A	0.02	0.33	2.63	48.96	44.49	1.70	1.86	0.26	0.22	0.414
CWF3B	0.02	0.26	2.17	41.55	50.58	2.41	3.01	0.25	0.39	0.596
CWF3C	0.00	0.66	2.87	41.01	51.79	1.85	1.82	0.25	0.34	0.455
CWF4	0.12	0.69	2.96	45.97	46.04	2.12	2.10	0.25	0.38	0.488
CS1A	0.05	0.36	0.96	38.31	57.93	0.72	1.68	0.24	0.36	0.246
CS1B	0.13	0.39	1.29	47.22	48.43	1.16	1.38	0.25	0.21	0.292
CS1C	0.00	0.42	1.46	47.82	47.11	1.46	1.74	0.25	0.20	0.334
CS2A	1.47	1.01	2.22	48.72	44.39	1.23	0.94	0.26	0.10	0.228
CS2B	1.88	1.88	3.18	44.76	43.99	1.61	2.71	0.26	0.45	0.772
CS2C	0.37	0.77	2.15	51.10	42.79	0.98	1.85	0.26	0.30	0.345
CS3A	0.30	0.95	6.78	59.94	30.54	0.39	1.10	0.30	0.30	0.242
CS3B	0.31	1.34	5.19	56.70	34.03	0.85	1.57	0.29	0.32	0.313
CS3C	3.15	3.34	8.89	58.49	24.09	0.66	1.39	0.33	0.29	0.179
SN3A	1.41	4.69	9.54	36.74	40.11	4.15	3.35	0.28	0.35	0.740
SN3B	2.05	6.99	12.35	38.08	33.33	3.30	3.89	0.33	0.45	0.815
SN3C	1.15	3.48	9.55	44.67	36.05	3.05	2.05	0.29	0.37	0.759
SN2A	0.66	2.67	8.93	55.09	27.58	2.67	2.41	0.30	0.31	0.808
SN2B	1.00	3.22	9.28	53.18	25.94	2.52	4.87	0.29	0.55	1.000
SN2C	1.09	2.30	7.21	52.59	30.32	3.33	3.16	0.28	0.40	0.633
SN1A	1.28	3.39	10.33	54.58	26.99	1.56	1.87	0.31	0.24	0.358
SN1B	0.84	3.66	11.55	56.98	24.90	0.82	1.26	0.32	0.22	0.245
SN1C	1.36	3.63	11.05	56.24	25.40	1.07	1.25	0.32	0.16	0.314
SNAA	1.06	4.20	11.93	57.53	23.67	0.61	1.00	0.34	0.06	0.398
SNAB	1.32	2.86	10.29	61.56	22.48	0.44	1.05	0.33	0.10	0.342

Sample	Gravel (%)	VCS (%)	CS (%)	MS (%)	FS (%)	VFS (%)	Mud (%)	Mean size (mm)	TOC (%)	COD (mg $O_2.g^{-1}$)
SNAC	1.50	4.06	12.54	64.43	15.89	0.44	1.14	0.37	0.16	0.391
SNBA	2.10	4.82	12.29	63.95	14.92	0.63	1.29	0.37	0.36	0.782
SNBB	1.94	4.52	11.60	57.60	21.86	1.21	1.27	0.34	0.28	0.638
SNBC	1.46	3.60	10.90	60.11	21.56	1.04	1.34	0.33	0.30	0.593
SWB2	1.22	3.14	11.09	62.17	19.34	1.83	1.22	0.33	0.28	0.933
SWB3A	1.74	3.83	11.54	55.56	24.06	1.74	1.53	0.33	0.32	0.836
SWB3B	1.34	5.31	12.99	55.97	21.31	1.48	1.61	0.35	0.21	0.712
SWB3C	1.64	4.15	11.56	55.22	24.11	1.73	1.60	0.33	0.33	1.200
SWB4	1.46	3.15	9.32	53.96	28.59	1.59	1.94	0.31	0.33	0.992
SWC2	1.12	3.48	9.62	56.45	25.07	2.28	1.98	0.31	0.25	1.070
SWC3A	1.04	2.56	9.33	61.31	22.05	0.91	2.79	0.32	0.40	1.040
SWC3B	0.69	2.21	7.53	57.34	27.86	1.78	2.60	0.30	0.48	1.350
SWC3C	0.64	2.53	8.94	56.86	27.20	1.71	2.13	0.30	0.49	0.758
SWC4	0.70	2.14	6.63	49.10	37.57	1.92	1.94	0.28	0.22	0.786
SWD2	1.15	3.24	9.36	56.40	25.37	2.84	1.64	0.31	0.36	0.988
SWD3A	0.57	1.42	5.61	52.56	33.52	2.35	3.97	0.28	0.74	2.030
SWD3B	1.03	2.14	8.54	60.11	23.93	1.97	2.29	0.31	0.50	1.690
SWD3C	0.55	2.26	9.01	52.00	31.09	2.34	2.75	0.29	0.94	1.340
SWD4	0.42	1.70	6.34	54.49	33.11	1.55	2.40	0.29	0.50	1.000
SWE2	0.72	2.79	12.04	57.98	22.37	2.20	1.90	0.32	0.52	0.826
SWE3A	0.63	3.03	13.34	55.47	23.89	1.33	2.31	0.33	0.59	1.010
SWE B	0.31	2.05	8.97	49.68	30.69	2.16	6.14	0.29	0.81	1.090
SWE3C	0.44	1.86	8.06	58.94	26.85	1.58	2.26	0.31	0.59	0.874
SWE4	0.28	2.46	9.83	57.60	25.34	1.74	2.76	0.31	0.46	0.949
SWF2	4.16	8.71	18.28	48.91	17.10	1.28	1.55	0.42	0.39	0.630
SWF3A	0.84	3.11	10.86	56.62	24.88	2.05	1.63	0.32	0.38	0.662
SWF3B	0.88	2.74	7.88	54.39	29.56	1.96	2.59	0.30	0.42	0.883
SWF3C	0.79	3.17	10.50	57.09	25.27	1.45	1.72	0.32	0.40	0.838
SWF4	1.08	3.17	10.40	57.37	25.25	1.02	1.71	0.32	0.23	0.516
SSAA	0.28	1.67	5.88	50.55	38.38	1.91	1.32	0.28	0.27	0.467
SSAB	0.44	1.99	8.92	52.39	33.47	1.16	1.65	0.29	0.11	0.462
SSAC	0.61	2.40	9.06	51.81	33.44	1.23	1.45	0.30	0.19	0.586
SSBA	0.72	2.16	6.89	49.89	37.82	1.29	1.22	0.29	0.28	0.439
SSBB	0.69	3.10	9.03	58.28	27.21	0.69	0.99	0.32	0.18	0.344
SSBC	1.07	4.69	10.11	54.90	27.31	0.75	1.17	0.32	0.19	0.353
SS1A	1.49	4.34	11.54	58.33	22.59	0.74	0.98	0.34	0.21	0.320
SS1B	1.42	3.27	8.92	60.86	23.17	1.13	1.22	0.32	0.22	0.336
SS1C	1.06	2.93	9.26	60.03	24.91	0.72	1.08	0.32	0.23	0.228
SS2A	4.41	4.94	9.08	53.13	25.90	1.65	0.88	0.34	0.10	0.340
SS2B	4.51	5.54	7.13	46.71	33.99	1.21	0.91	0.32	0.09	0.292
SS2C	8.39	7.05	7.20	44.78	29.79	1.47	1.32	0.38	0.10	0.340

Appendix 4.6.2. Faecal indicator bacteria counts in sediment, sea urchin fertilisation success following exposure to porewater, and ammonia concentrations in porewater of sediment collected for the 2011 survey of the Durban outfalls monitoring programme.

Sample	Faecal coliforms cfu/50 g	Faecal streptococci cfu/50 g	Sea Urchin Average Fertilisation Success (%)	Sea Urchin Fertilisation Standard Deviation	Total Ammonia-N ($mg.l^{-1}$)
CN1	63	627	94.50	0.58	1.40
CWB3	263	876	90.00	0.82	2.54
CWC3	43308	1299	86.75	0.96	4.37
CWD3	3755	626	94.50	0.58	1.74
CWE3	180	180	92.50	0.58	No data
CWF3	289	1620	92.50	0.58	1.26
CS1	134	1340	88.75	0.96	No data
CS2	199	20	91.75	0.96	1.00
CS3	928	232	8.75	0.96	0.354
SN3	1084	217	79.75	1.71	5.11
SN2	3098	194	90.75	0.96	2.10
SN1	2037	1811	75.00	1.41	3.33
SNA	1901	1228	80.75	1.71	1.51
SNB	2801	1027	73.25	1.26	7.54
SWB3	2644	1102	1.50	0.58	5.82
SWC3	3710	1893	1.25	0.50	8.92
SWD3	3148	1023	0.00	0.00	20.30
SWE3	22682	9279	0.00	0.00	12.76
SWF3	477	3819	2.25	0.50	10.30
SSA	488	206	75.25	1.26	6.38
SSB	1500	262	85.25	0.96	4.16
SS1	369	225	65.75	0.96	9.03
SS2	270	74	81.25	1.50	3.53

Appendix 4.6.3. Metal concentrations ($mg.g^{-1}$ for aluminium and iron, $\mu g.g^{-1}$ for all other metals; dry weight) in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. Al = aluminium, Fe = iron, As = arsenic, Cd = cadmium, Co = cobalt, Cu = copper, Cr = chromium, Hg = mercury, Mn = manganese, Ni = nickel, Pb = lead, V = vanadium, Zn = zinc, < = concentration below method detection limit.

Sample	Al	Fe	As	Cd	Co	Cu	Cr	Hg	Mn	Ni	Pb	V	Zn
CN1A	2.42	4.01	3.51	0.165	1.69	1.56	10.62	0.020	57.38	2.31	6.03	6.29	7.19
CN1B	2.38	3.88	4.14	0.274	1.55	1.77	9.88	0.024	53.08	1.96	6.44	6.27	6.61
CN1C	2.35	3.71	3.27	0.190	1.73	1.50	11.28	0.029	43.54	1.72	7.65	7.95	7.71
CWB2	2.78	4.52	4.81	0.054	1.63	1.28	11.46	0.019	52.56	2.86	1.80	12.28	5.50
CWB3A	4.38	5.31	5.22	0.073	2.40	2.43	18.61	0.041	56.63	2.86	3.55	12.34	7.95
CWB3B	3.93	5.37	5.70	0.166	1.55	2.51	17.52	0.030	59.10	4.45	3.71	12.54	10.26
CWB3C	4.16	5.45	4.47	0.164	2.19	2.34	18.26	0.038	62.46	3.09	4.32	10.26	9.44
CWB4	3.59	5.36	4.86	0.228	2.10	3.62	17.48	0.037	55.59	3.90	11.86	6.27	12.21
CWC2	2.63	3.65	4.45	0.062	2.50	1.31	9.65	0.011	42.61	3.12	1.66	7.48	4.92
CWC3A	3.04	3.79	4.92	0.133	1.72	2.14	12.01	0.021	51.35	1.36	1.17	9.66	12.69
CWC3B	3.22	4.91	3.71	0.310	1.91	2.45	12.76	0.027	51.19	2.27	3.32	7.92	7.65
CWC3C	2.85	4.20	4.26	0.487	1.88	1.55	12.72	0.034	44.71	2.25	1.77	8.10	5.86
CWC4	2.87	4.35	6.45	0.123	1.57	2.26	12.53	0.016	50.23	2.13	1.90	7.92	6.75
CWD2	3.16	4.40	3.16	0.274	1.58	1.87	12.29	0.012	52.56	2.18	3.44	9.19	6.53
CWD3A	3.27	4.55	3.37	0.134	1.27	1.69	13.42	0.023	61.42	1.72	2.93	10.08	8.09
CWD3B	3.19	4.48	4.54	0.219	1.29	2.22	13.25	0.021	50.84	1.75	4.57	7.63	6.46
CWD3C	2.68	4.20	5.45	0.177	1.31	1.05	11.03	0.011	47.43	1.50	4.82	10.81	4.62
CWD4	3.67	5.30	4.88	0.413	2.17	6.46	17.02	0.015	58.75	1.35	12.38	8.74	24.19
CWE2	2.56	4.04	6.17	0.094	1.97	1.26	12.60	0.030	51.05	3.56	1.97	8.59	5.34
CWE3A	3.07	4.48	5.40	0.396	1.35	37.10	11.28	0.012	58.09	2.99	1.89	8.81	4.82
CWE3B	3.19	4.71	5.00	0.160	1.20	2.17	12.25	0.031	67.24	2.18	1.80	8.87	7.24
CWE3C	4.60	6.21	3.76	0.094	2.33	3.56	12.40	0.038	69.83	1.70	5.09	9.88	10.72
CWE4	3.87	5.09	4.27	0.197	1.56	1.74	19.35	0.033	59.93	3.30	4.51	10.26	8.00
CWF2	3.20	5.04	5.49	0.111	1.70	1.81	12.55	0.030	57.84	3.90	2.31	7.16	7.58
CWF3A	3.08	4.28	8.10	0.102	1.88	1.75	10.54	0.027	50.61	3.07	2.15	7.65	6.76
CWF3B	2.69	4.25	6.72	0.243	1.32	2.23	10.48	0.026	45.20	1.03	1.33	9.73	5.46
CWF3C	2.95	5.20	6.98	0.175	1.59	1.17	13.67	0.010	57.57	3.61	2.00	9.52	7.20
CWF4	2.99	5.29	7.67	0.081	1.62	1.08	13.87	0.016	57.30	2.91	2.86	6.19	6.85
CS1A	2.27	3.81	3.32	0.073	1.79	3.61	10.73	0.022	60.84	1.47	1.96	6.40	12.72
CS1B	2.86	3.95	6.63	0.167	1.29	1.28	10.35	0.016	75.77	3.42	2.24	6.75	5.22
CS1C	2.64	4.40	4.36	0.115	1.28	1.13	11.75	0.027	94.15	1.86	1.96	6.10	6.36
CS2A	2.22	3.52	4.44	0.082	1.66	1.18	9.60	0.012	58.96	1.73	2.17	9.06	4.29
CS2B	3.41	4.80	5.37	3.972	1.96	2.01	12.73	0.011	85.38	2.13	2.35	6.61	7.45
CS2C	2.39	3.98	3.90	0.062	1.90	1.16	10.62	<0.01	64.67	1.64	2.19	4.99	4.41
CS3A	1.96	2.73	3.20	0.145	1.75	1.49	7.20	0.012	86.62	2.09	1.88	6.49	3.56
CS3B	2.34	3.25	4.11	0.247	1.98	1.06	9.50	0.011	55.30	1.20	1.85	5.98	4.54
CS3C	2.19	3.04	4.49	0.221	2.11	1.11	8.30	<0.01	55.42	1.38	1.77	8.36	3.35
SN3A	3.53	5.79	4.05	0.247	2.15	2.45	17.67	0.045	64.77	2.31	3.12	7.82	10.09
SN3B	3.54	5.67	4.32	0.088	2.18	2.24	15.64	0.017	65.78	2.73	1.42	8.04	9.24
SN3C	3.29	5.50	4.29	0.233	1.78	2.26	16.43	0.013	58.85	3.07	3.45	7.41	9.22
SN2A	3.41	4.82	4.25	0.062	1.39	1.88	13.36	0.014	60.72	1.97	2.63	7.56	6.99
SN2B	3.79	5.55	4.58	0.139	2.29	2.73	16.76	<0.01	73.03	2.47	3.57	7.21	8.55
SN2C	2.43	3.85	3.43	0.080	1.80	1.39	10.25	0.015	56.25	2.07	3.14	9.23	5.48
SN1A	3.04	4.68	3.34	0.273	1.89	1.67	13.32	0.012	63.35	1.52	2.59	8.10	7.16
SN1B	2.54	4.25	3.60	0.162	2.04	1.15	11.51	<0.01	65.76	2.07	2.69	9.57	5.40
SN1C	2.86	4.52	3.61	0.078	1.95	1.51	13.86	0.014	64.80	1.23	1.69	7.41	6.09
SNAA	2.32	3.84	2.79	0.178	1.68	1.01	11.03	0.015	66.92	2.01	3.35	8.01	4.08
SNAB	2.64	4.21	2.58	0.252	1.86	1.31	12.06	<0.01	65.68	1.20	4.71	7.29	4.77
SNAC	2.26	3.79	3.87	0.109	2.12	0.98	10.94	0.012	64.62	1.19	3.34	8.97	4.12
SNBA	2.78	4.39	3.11	0.125	1.23	1.82	13.07	0.017	71.49	1.46	2.34	8.87	7.07
SNBB	2.64	4.30	3.09	0.074	2.24	1.96	12.88	0.005	82.39	1.33	2.83	8.46	6.02

Sample	Al	Fe	As	Cd	Co	Cu	Cr	Hg	Mn	Ni	Pb	V	Zn
SNBC	2.93	4.26	3.38	0.218	1.73	2.15	12.01	0.012	66.70	1.44	2.86	7.27	6.73
SWB2	2.95	4.43	7.54	0.065	1.81	2.46	13.73	0.011	87.70	2.16	3.02	9.50	7.65
SWB3A	2.49	3.84	4.85	0.089	1.97	6.20	13.88	0.038	55.55	1.79	1.05	7.78	10.02
SWB3B	2.93	4.77	6.92	0.202	2.31	3.53	17.92	0.045	73.14	2.73	5.40	7.23	17.31
SWB3C	2.95	4.51	6.25	0.317	1.33	4.61	16.09	0.037	72.13	1.33	1.15	8.36	13.98
SWB4	2.93	4.95	5.79	0.188	2.00	3.72	16.98	0.022	70.53	2.35	1.32	8.31	9.35
SWC2	3.01	4.79	6.87	0.113	1.67	4.97	16.77	0.026	71.35	1.78	1.15	8.85	13.60
SWC3A	2.33	3.61	4.61	0.244	1.97	7.87	13.13	0.044	52.12	1.56	1.63	7.66	11.19
SWC3B	3.04	4.65	8.77	0.195	1.52	6.44	17.36	0.040	71.34	3.11	3.08	6.17	15.24
SWC3C	2.75	3.16	5.41	0.289	1.33	5.55	10.67	0.033	50.82	1.06	2.46	12.80	9.17
SWC4	3.00	4.83	7.05	0.165	1.40	3.75	17.51	0.033	61.97	3.87	2.38	11.59	9.82
SWD2	2.77	4.49	4.55	0.096	1.69	3.91	14.94	0.033	65.48	3.20	1.27	7.91	11.38
SWD3A	2.99	3.78	5.75	0.163	1.33	15.73	15.30	0.048	69.00	4.72	2.71	8.25	35.18
SWD3B	2.80	3.65	5.56	0.368	1.45	5.29	11.96	0.074	43.71	3.38	2.37	7.57	16.92
SWD3C	2.99	3.38	4.01	0.085	1.30	6.59	11.80	0.052	35.24	4.69	6.84	7.73	26.66
SWD4	2.72	3.99	5.10	0.159	1.29	2.37	13.79	0.036	52.30	1.28	2.96	10.98	11.59
SWE2	2.62	4.24	5.82	0.103	1.68	2.44	14.94	0.047	51.16	2.67	1.37	9.00	11.34
SWE3A	2.87	4.03	8.53	0.495	1.78	5.73	12.86	0.142	51.99	4.79	2.01	8.82	19.38
SWE B	3.15	3.83	6.00	0.290	1.54	9.32	13.29	0.101	49.30	5.30	6.86	8.28	33.11
SWE3C	3.00	3.63	4.09	0.373	1.68	3.47	12.24	0.081	44.95	4.90	1.75	13.04	19.53
SWE4	3.48	4.88	5.19	0.054	1.68	10.94	17.88	0.044	68.01	2.68	2.03	11.31	16.80
SWF2	2.61	4.06	5.63	0.072	1.68	1.67	13.21	0.016	68.28	2.73	2.38	8.17	8.09
SWF3A	2.80	3.50	7.37	0.382	1.57	4.57	10.81	0.054	45.66	2.68	2.53	10.18	10.78
SWF3B	3.40	4.24	7.60	0.306	2.05	2.71	13.01	0.020	48.86	3.67	3.07	8.56	15.75
SWF3C	2.76	3.86	5.53	0.250	1.45	8.93	11.16	0.025	44.96	1.86	3.07	10.98	10.29
SWF4	2.57	4.34	6.16	0.347	1.36	1.63	13.70	0.018	66.17	2.65	1.29	9.69	8.19
SSAA	2.78	4.51	4.39	0.211	1.27	1.77	14.20	<0.01	64.25	1.26	2.10	9.89	8.27
SSAB	3.02	4.87	3.47	0.047	1.67	3.51	13.80	<0.01	66.94	1.64	2.87	8.64	8.66
SSAC	2.59	4.35	5.73	0.378	1.87	2.43	12.78	0.025	59.11	1.93	2.03	9.91	9.60
SSBA	2.96	5.07	5.25	0.129	1.50	3.20	13.24	0.017	67.54	4.09	2.45	9.27	8.13
SSBB	2.62	4.84	7.43	0.192	1.42	2.81	12.87	0.018	67.63	1.91	2.22	10.73	7.87
SSBC	3.14	5.52	8.93	0.177	1.57	1.94	13.71	0.038	75.52	3.20	1.79	9.73	8.90
SS1A	3.28	4.85	6.07	0.289	1.41	1.50	12.95	0.011	93.37	1.77	2.04	9.24	7.84
SS1B	2.67	4.51	5.97	0.141	1.22	1.26	12.30	0.017	85.38	3.38	2.76	9.31	7.33
SS1C	2.69	4.85	6.76	0.022	2.01	1.13	13.28	<0.01	83.04	1.39	4.15	9.29	7.67
SS2A	3.04	4.61	5.99	0.239	1.95	1.33	11.56	<0.01	86.41	1.08	2.72	7.71	6.68
SS2B	2.61	4.16	5.75	0.080	1.83	1.14	11.04	<0.01	77.51	1.71	3.60	10.55	5.65
SS2C	2.94	5.10	7.41	0.016	1.78	2.11	13.45	0.020	74.79	3.60	2.54	11.13	6.73

Appendix 4.6.4. Organic chemical pollutant concentrations ($mg.kg^{-1}$ dry weight) in sediment collected for the 2011 survey of the Durban outfalls monitoring programme. < = concentration below method detection limit.

Class	Analyte	CN1	CWB3	CWC3	CWD3	CWE3	CWF3	CS1	CS2	CS3	SN3	SN2	SN1
Organochlorine pesticides	alpha-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	beta-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	gamma-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	delta-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Hexachlorobenzene	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Heptachlor	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Heptachlorepoxide (cis)	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Heptachlorepoxide (trans)	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Hexachlorbutadiene	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Aldrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Dieldrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Endrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Isodrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Telodrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	alpha-Endosulfan	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	beta-Endosulfan	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	alpha-Endosulfan sulphate	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	alpha-Chlordane	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	gamma-Chlordane	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	o'p'-DDT	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	p'p'-DDT	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	o'p'-DDE	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	p'p'-DDE	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	o'p'-DDD	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	p'p'-DDD	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
Polychlorinated biphenyls	PCB 28	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 52	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 101	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 118	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 138/163	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 153	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 180	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB (7) sum	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070
Polycyclic Aromatic Hydrocarbons	Naphthalene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010

Class	Analyte	CN1	CWB3	CWC3	CWD3	CWE3	CWF3	CS1	CS2	CS3	SN3	SN2	SN1
	Acenaphthylene	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050
	Acenaphthene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Fluorene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Phenanthrene	0.023	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Anthracene	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Fluoranthene	0.04	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Pyrene	0.029	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Benz(a)anthracene	0.024	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Chrysene	0.024	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(b)fluoranthene	0.028	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(k)fluoranthene	0.011	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(a)pyrene	0.024	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Dibenzo(ah)anthracene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(ghi)perylene	0.015	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Indeno(123-cd)pyrene	<0.010	<0.010	<0.010	<0.010	<0.010	0.014	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010

Appendix 4.6.4. Continued.

Class	Analyte	SNA	SNBB	SWB3	SWC3	SWD3	SWE3	SWF3	SSA	SSB	SS1	SS2
Organochlorine pesticides	alpha-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	beta-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	gamma-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	delta-HCH	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Hexachlorobenzene	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Heptachlor	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Heptachlorepoxide (cis)	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Heptachlorepoxide (trans)	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Hexachlorbutadiene	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Aldrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Dieldrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Endrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Isodrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Telodrin	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	alpha-Endosulfan	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	beta-Endosulfan	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	alpha-Endosulfan sulphate	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	alpha-Chlordane	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	gamma-Chlordane	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010

Class	Analyte	SNA	SNBB	SWB3	SWC3	SWD3	SWE3	SWF3	SSA	SSB	SS1	SS2
	o'p'-DDT	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	p'p'-DDT	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	o'p'-DDE	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	p'p'-DDE	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	o'p'-DDD	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	p'p'-DDD	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
Polychlorinated biphenyls	PCB 28	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 52	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 101	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 118	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 138/163	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 153	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB 180	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	PCB (7) sum	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070	<0.0070
Polycyclic Aromatic Hydrocarbons	Naphthalene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Acenaphthylene	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050	<0.050
	Acenaphthene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Fluorene	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Phenanthrene	<0.010	<0.010	<0.010	<0.010	<0.010	0.058	<0.010	<0.010	<0.010	<0.010	<0.010
	Anthracene	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	0.015	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Fluoranthene	<0.010	<0.010	<0.010	<0.010	0.020	0.210	<0.010	<0.010	<0.010	<0.010	<0.010
	Pyrene	<0.010	<0.010	<0.010	<0.010	0.017	0.100	<0.010	<0.010	<0.010	<0.010	<0.010
	Benz(a)anthracene	<0.010	<0.010	<0.010	<0.010	<0.010	0.091	<0.010	<0.010	<0.010	<0.010	<0.010
	Chrysene	<0.010	<0.010	<0.010	<0.010	0.010	0.110	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(b)fluoranthene	<0.010	<0.010	<0.010	<0.010	0.020	0.110	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(k)fluoranthene	<0.010	<0.010	<0.010	<0.010	<0.010	0.045	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(a)pyrene	<0.010	<0.010	<0.010	<0.010	<0.010	0.086	<0.010	<0.010	<0.010	<0.010	<0.010
	Dibenzo(ah)anthracene	<0.010	<0.010	<0.010	<0.010	<0.010	0.011	<0.010	<0.010	<0.010	<0.010	<0.010
	Benzo(ghi)perylene	<0.010	<0.010	<0.010	<0.010	<0.010	0.093	<0.010	<0.010	<0.010	<0.010	<0.010
	Indeno(123-cd)pyrene	<0.010	<0.010	<0.010	<0.010	<0.010	0.098	<0.010	<0.010	<0.010	<0.010	<0.010

Chapter 5
Benthic Macrofauna

5.1. Introduction

A critical end-point of any outfall monitoring programme is to determine whether effluent discharge is adversely impacting the ecology of the receiving environment. Little relevance can be attached to a particular degree of contamination of water and sediment unless it can be evaluated within a biological context. This context might be derived from an empirically established relationship between the degree of contamination and biological response, or through toxicity testing, which, in all its levels of complexity, provides a powerful tool for predicting potential biological response. Perhaps the most widely used tool for measuring biological response in marine outfall (and other environmental) monitoring programmes, however, is the evaluation of benthic macrofaunal community structure and composition. Indeed, such monitoring forms a common thread through most outfall monitoring programmes worldwide.

The benthic macrofauna, which includes invertebrate species that dwell in or on the seabed and are retained by a 1 mm mesh size sieve, has emerged as the most appropriate group of organisms for assessing the impact of effluent discharge through marine outfalls. These fauna have been widely exploited as a proxy measure of environmental impact (e.g. Pearson and Rosenberg 1978, Diener et al. 1995). In contrast to pelagic fauna (such as fish and plankton), which can move in and out of an area, thereby avoiding temporarily contaminated waters, and are also affected by currents, benthic macrofauna, by virtue of their relatively sedentary lifestyle, must cope with prevailing conditions. Benthic macrofauna tend to have generation times that extend over months or years and are, therefore, particularly useful for integrating the effects of impact over the relatively long inter-survey timeframe of one year adopted for the Durban outfalls monitoring programme. Benthic macrofauna typically include a wide variety of taxa with varying sensitivities to particular impacts. Response to contamination is, therefore, species specific, but inevitably species response is reflected at the community level. Chronic exposure to contamination may cause sensitive species to die and allow more tolerant, opportunistic species to proliferate. The net effect in such cases is a skewing of community composition and structure. This can be interpreted to reflect the general state of the environment. Pollution impacts then are reflected by shifts in the abundance of component species, reduction in diversity, or a relative proliferation of pollution tolerant and opportunistic species.

The KwaZulu-Natal coastal shelf supports diverse communities of benthic macrofauna (McClurg 1988). These organisms are essential members of the marine ecosystem and serve not only in providing a food source for higher trophic levels but also in recycling nutrients. The structure of marine benthic macrofaunal communities is influenced by many natural factors. These include abiotic factors, such as sediment (e.g. grain size) and water conditions (e.g. salinity, temperature, current velocity), and biotic factors (e.g. predation, competition, availability of food). Anthropogenic activities such as pollution, can play a significant role in modifying the structure of the marine benthic macrofaunal communities. In assessing the environmental impact of effluent discharge through outfalls it is important to make a clear distinction between natural and anthropogenically induced differences. This is usually achieved through comparison of potentially impacted communities with fauna reflective of background or reference conditions, as displayed in samples taken beyond the influence of the effluent discharge. Spatial heterogeneity in benthic macrofaunal distributions may, however, be naturally high, and a major challenge in marine ecological impact assessment is to identify anthropogenic disturbance against an inherently ‘noisy’ natural background.

While the ideal situation would be to have no impact, this is clearly an unrealistic expectation with outfalls that discharge large volumes of effluent on a daily basis. Outfall design usually presupposes that there will be an area around the point of effluent discharge (i.e. diffuser section) where there will

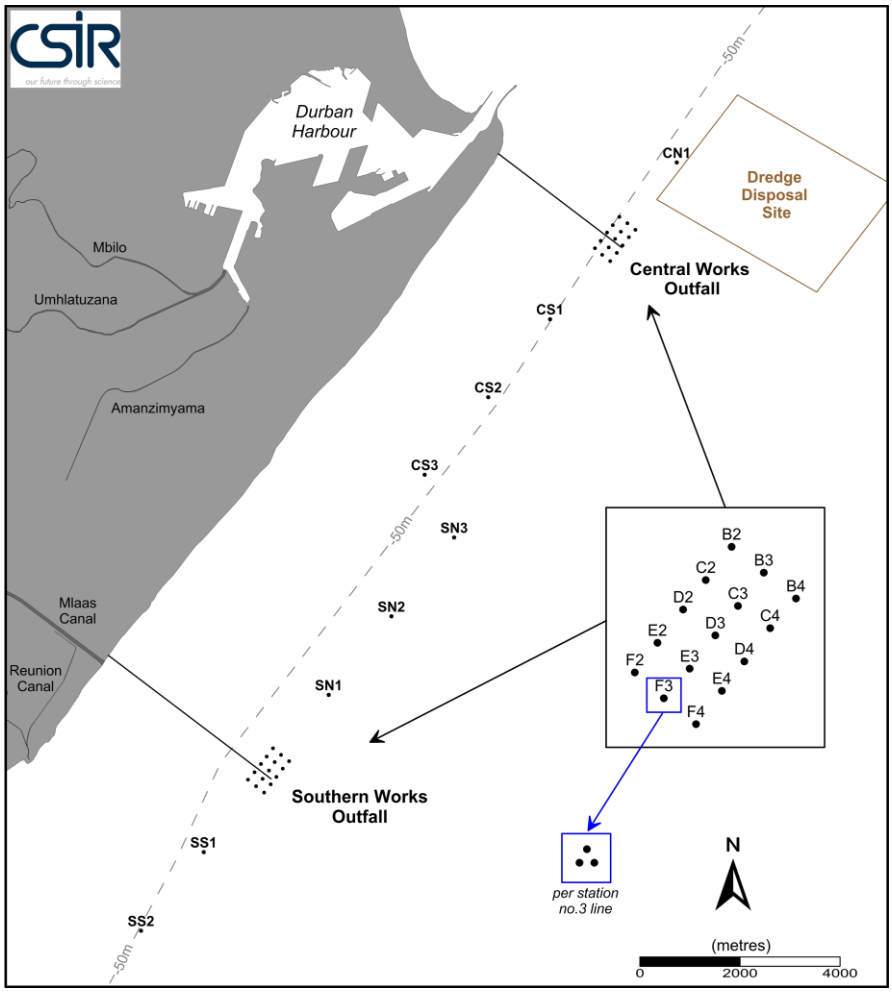

Figure 5.1. Map illustrating the positions where benthic macrofaunal communities were sampled for the 2011 survey of the Durban outfalls monitoring programme.

inevitably be localised high concentrations of contaminants during the initial stages of effluent dilution. This has been variously described as the zone of initial dilution and the sacrificial zone, and, for a buoyant effluent, is usually taken to include an envelope of seawater extending from sea surface to seabed and laterally to a position determined as two to three times the water depth at the point of discharge. The zone of initial dilution for a discharge in 60 metres would, therefore, extend 120 to 180 metres in all directions from the point of effluent discharge. The concept of a zone of initial dilution has important practical implications for monitoring surveys that employ water quality guidelines since it allows direct measurement to confirm that adequate dilution is being achieved within a prescribed area. Ideally, one might expect the zone of initial dilution to include the seabed. However, this is seldom the case as an array of additional factors comes into play once material has settled on the seabed or contaminants have become bound to particulates. Dispersion, rather than simple dilution, becomes the main driving force. Also, the techniques used to monitor impact become less clear-cut. Instead of using clearly defined, and widely accepted, water quality guidelines, the process becomes more reliant on measures that are influenced by the complex interaction of physical, chemical and biological processes.

The rationale adopted in the Durban outfalls monitoring programme is that, while a totally pristine seabed is desirable, moderate temporary aberrations are expected near the discharge. These serve as a trigger for management intervention. However, in common with most monitoring programmes worldwide, the Durban outfalls monitoring programme has not defined an *a priori* level at which an impact becomes 'unacceptable' and requires management intervention. Acceptability is a matter of opinion and depends largely on one's frame of reference. While it is important from a democratic perspective that multiple opinions are sought, it is equally important

that decisions are made on a rational and informed basis rather than on emotional perceptions.

From the early days of effluent discharge through the Durban outfalls, benthic macrofaunal surveys have revealed, as would be expected, instances of moderate organic enrichment of sediment in the immediate vicinities of both outfalls. In recent years there has been evidence of increasing organic enrichment in the vicinity of the Southern Works outfall. The impact of this increasing enrichment has most clearly manifested in a reduction in species diversity and increased abundance of capitellid polychaetes (opportunist marine worms associated with high organic loads). From an ecological perspective, this does not appear to constitute a serious threat, particularly since the effects are localised and transitory. However, over the past decade there has been strong evidence of an upward trend in the number of stations near the Southern Works outfall discharge that manifest this impact.

This chapter analyses and discusses the results of benthic macrofaunal surveys conducted on 24 and 25 May 2011. The major objectives are to assess the impact of effluent discharge on benthic macrofauna communities and to analyse temporal trends.

5.2. Materials and Methods

5.2.1. Fieldwork and Laboratory Analyses

A Day grab that sampled a surface area of 0.25 m^2 was used to collect sediment from 15 stations arranged in a grid-like manner spanning the diffuser sections of the Central Works and Southern Works outfalls (Figure 5.1). The stations cover an area of about 0.32 km^2, with stations situated about 200 m apart in offshore and alongshore directions. The most distant stations of each grid (B2, B3 and B4 in Figure 5.1) were situated about 500 m from the diffuser section of each outfall. Two additional sediment samples were collected at each station along the central line of the grid running parallel to the shoreline (the so-called 3 line), to provide a total of three replicate samples. A total of 89 successful grab samples was required to satisfy the sampling design. For discussion purposes, stations of the grid at which replicate samples were collected are referred to as sites, while the entire grids, either including or excluding non-replicated stations, are referred to as the Central Works or Southern Works outfall sites. Three sediment samples were also collected at each of the 13 reference sites situated to the north-northeast and south-southwest of the outfalls. There were four reference sites for the Central Works outfall and nine for the Southern Works outfall. Further for discussion purposes, sites of the sampling grids and the reference sites for each outfall are referred to as falling on the Central Works or Southern Works lines of sites.

The Day grab used to collect sediment has hinged top screens that prevent disturbance and washout of sediment during retrieval. Following retrieval of the grab, an inspection of the contents was made. A minimum of 5 cm of undisturbed and level sediment inside the grab was required for the grab to be accepted. If these conditions were not met the contents were discarded and the grab was again deployed. For successful grabs, water overlaying sediment in the grab was siphoned off, the contents photographed, and the sediment inspected for its texture, colour, aroma and the presence of obviously anomalous matter (e.g. tomato seeds, cigarette butts). The findings of these observations were noted on field data sheets. For most grabs, approximately 150 cm^3 of surface sediment was removed for physical and chemical analyses. The remainder of the sediment was then washed through 1 mm mesh size sieve to retain the macrofauna and coarser debris. This was immediately preserved in 5% formaldehyde. The sampling vessel steamed back to the position before the next deployment of the grab at stations/sites where replicate samples were collected or where the grabs contents were deemed unacceptable. The grab was rinsed with site water and scrubbed with a hard brush before each deployment.

In the laboratory, samples were first eluted with freshwater through a 0.25 mm mesh size sieve to yield the lighter organisms and then microscopically examined to remove (with fine forceps) denser and more cryptic material. The composite fauna for each station was preserved in 70% ethanol and subsequently identified to the lowest level of taxonomic resolution practicable and enumerated (see Appendices 5.6.1 and 5.6.2).

5.2.2. Data Analysis

PRIMER v6 software (Clarke and Warwick 2001),

which has achieved international acceptance as a tool for analysing and interpreting ecological community data, was used to delineate trends in survey results. The following PRIMER routines were used to analyse the data. Where appropriate data were log (X + 1) transformed prior to analysis.

DIVERSE: Calculates a range of community variables and diversity indices for each sample. These are key ecological measures that are useful as indicators of the wellbeing of an ecosystem (Magurran 1988). They included, for the purposes of this study, total number of taxa, total number of individuals, species richness (Margalef) and diversity (Shannon-Weiner). These community variables and diversity indices can be subjected to statistical analysis to reveal spatial and temporal patterns of ecosystem health. In this case, average index values were compared amongst sites using one-way Analysis of Variance (ANOVA), followed where appropriate by a Tukey multiple comparison test (Zar 1996).

CLUSTER: A multivariate routine that measures the relative similarity of each sample within a group of samples. The end product is a dendrogram that displays hierarchical relationships, in terms of similarity, between each faunal sample within the group. It is particularly useful in delineating clusters of similar stations. Counts were square-root transformed to reduce the dominance of disproportionately high counts. The Bray-Curtis coefficient was used as a measure if similarity.

MDS: A multivariate routine that is related and complementary to CLUSTER, and which aims to depict similarities amongst samples within a spatial framework rather than a hierarchy. The product is an ordination (or scatterplot), which usually depicts the relationships amongst samples within a two-dimensional framework. Counts were square-root transformed to reduce the dominance of disproportionately high counts. The Bray-Curtis coefficient was used as a measure if similarity.

ANOSIM: A routine that allows statistical differences between groupings to be determined. To avoid circular arguments it should only be applied to groupings that are defined *a priori* and not to the groupings that are revealed by CLUSTER or MDS.

SIMPER: A routine that is used to examine the groupings revealed by CLUSTER, MDS and ANOSIM, and indicate which elements of the fauna are most responsible for defining the faunal character of those groupings. It is particularly useful for identifying faunal markers (indicator species) that may be indicative of impact.

BEST: Perhaps the most important routine in that it links biological patterns with physical and chemical measurements. By cross correlating the data sets, this routine defines which of the physical and chemical variables are most likely 'driving' the biological trends. It provides a vital link towards resolving cause and effect.

5.3. Results and Discussion

5.3.1. Univariate Analyses

Univariate analysis of community structure essentially collapses the data for each sample into a single index, for example a diversity index, and then makes use of face-value comparisons and statistical procedures to identify trends. Apart from the obvious measures, such as numbers of taxa and total counts, appropriate univariate measures of community structure include species richness, diversity and evenness. These measures provide insight into community composition. Conventional wisdom in assessing marine macrobenthic communities is that a 'healthy' community is characterised by high diversity and an even spread of numbers amongst species. A superabundance of one species in combination with reduced diversity often indicates that the community is stressed.

Figure 5.2 presents the average index values for sites falling on the Central Works or Southern Works lines of sites. Community metrics for the Central Works outfall show some variation across the study area, and in some cases the differences between sites are statistically significant ($p < 0.05$). However, these differences were not consistent between reference and outfall sites and indeed give little evidence of a pollution impact over the greater Central Works outfall site. This trend has been consistent over the last three surveys (and indeed over the longer history of the Durban outfalls monitoring programme). A noted exception to the above was the lower Pielou's Evenness Index at several sites (CWD3, CWE3, CWF3). Indeed, these sites were characterised by an abundance of capetellid

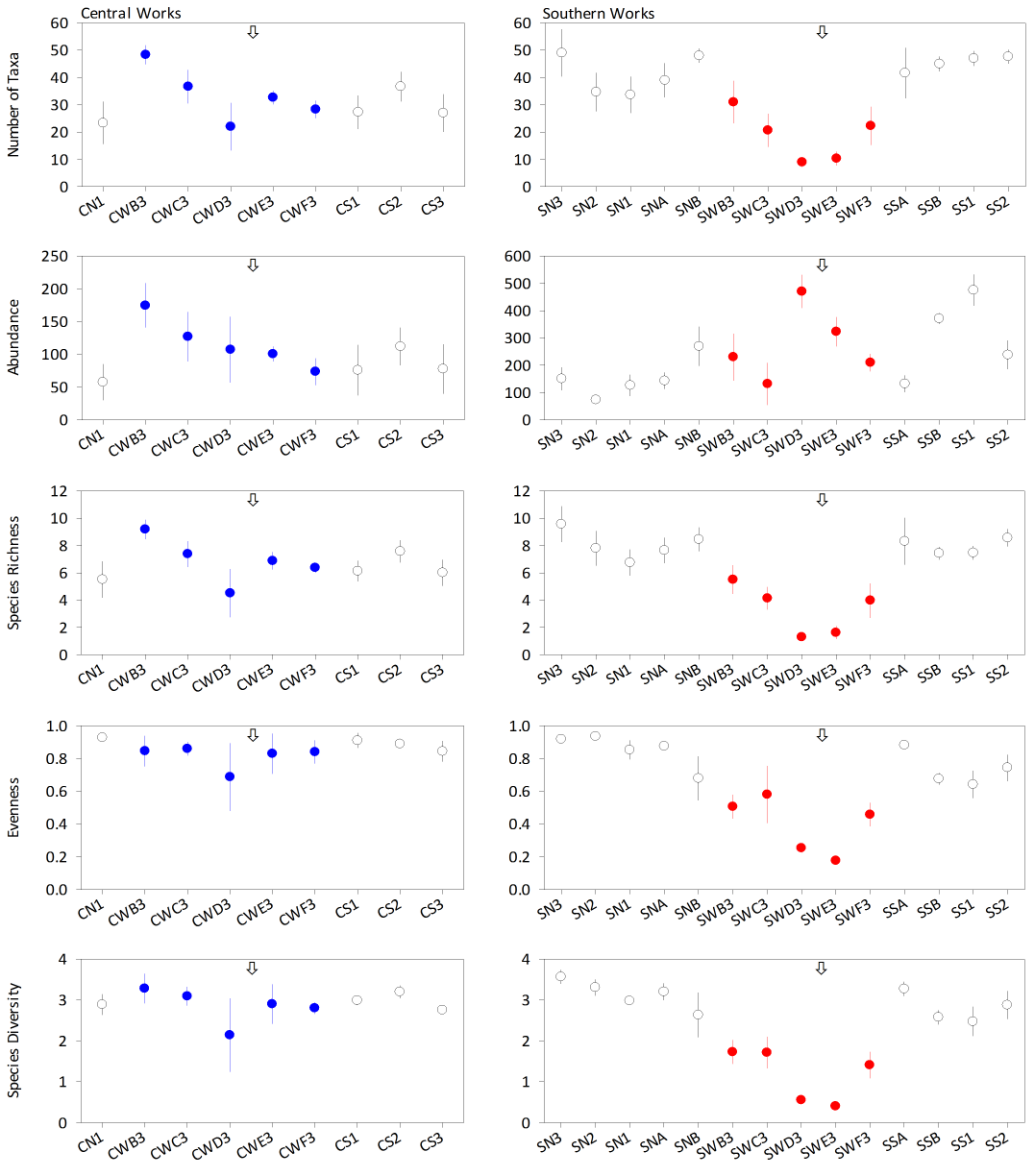

Figure 5.2. Indices of macrobenthic community structure for the 2011 survey of the Durban outfalls monitoring programme. Arrows denote the relative positions of the outfall diffusers. Solid symbols denote samples collected at sites on the so-called 3 line of the grid of stations spanning the diffuser section of each outfall, while open symbols denote samples collected at reference sites.

polychaetes relative to the reference sites, and relative to the 2009 and 2010 surveys (see later, Figure 5.4). Capetellid polychaetes are a group of marine worms that are tolerant of pollution and have an affinity for sediment with a high total organic content. Their abundance at Southern Works outfall sites has been recognised as a result of the accumulation of particulate organic matter in sediment. Sampling returned low to moderate numbers of these worms in several Central Works outfall samples. At these levels and with no indication of persistence, this is presently of little concern but requires close attention in future surveys.

In contrast to the Central Works outfall, marked variability was noted at Southern Works outfall sites, with statistically significant differences ($p < 0.05$) between reference and outfall sites (Figure 5.2). In all cases, the differences involved outfall sites showing symptoms of community degradation, with lower numbers of taxa, species richness and diversity. Higher abundance of organisms at Southern Works outfall sites was the result of a superabundance of capetellid polychaetes. This provides a clear indication of an effluent discharge induced impact. At the reference sites, benthic community metrics reflected highest numbers of taxa, species richness and diversity, suggesting no

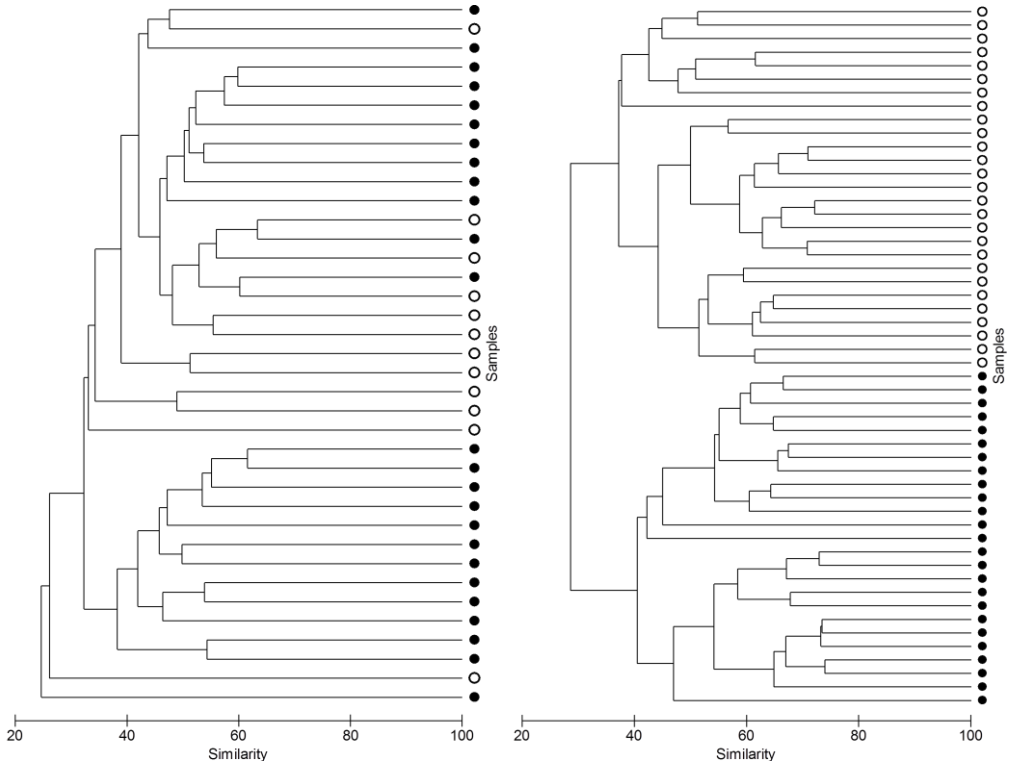

Figure 5.3. Cluster analysis dendrograms of macrobenthic samples taken from all outfall and reference stations/sites for the Central Works (left) and Southern Works outfall (right) for the 2011 survey of the Durban outfalls monitoring programme. Open symbols represent reference samples and closed symbols outfall samples.

impairment attributable to the outfall. This trend has been remarkably consistent over the recent years, as reflected in the 2009 and 2010 data (see later, Figure 5.4).

5.3.2. Multivariate Analyses

Multivariate analysis presents a powerful tool for developing an understanding of ecological impact because it allows a combined analysis of biological community characteristics and univariate physico-chemical measures.

Dendrograms generated by CLUSTER analysis of the benthic macrofauna data for Central Works and Southern Works are presented in Figure 5.3. In both dendrograms, samples taken at reference sites separate from those taken at outfall sites. Significant differences between outfall and reference sites were confirmed by ANOSIM (Global R = 0.473, p <0.01 for Central Works outfall; Global R = 0.695, p <0.01 for Southern Works outfall). The higher Global R statistic and the cluster plots themselves indicate that differences are much more pronounced at the Southern Works outfall, where two distinct clusters of samples were indentified (Figure 5.3). The clusters represent exclusively outfall and reference site samples. The dendrogram for the Central Works outfall is less distinct. Although there are two main clusters, one of the clusters contains a mixture of outfall and reference site samples (Figure 5.3).

SIMPER analyses indicated that dissimilarities between reference and outfall samples for the Central Works outfall were the result of subtle differences in a wide range of taxa. A relative abundance of *Capitella* polychaetes was, however, the second highest contributing factor to differences noted between outfall and references sites. For the Southern Works outfall, samples were strongly dominated by capitellid polychaetes and the relative abundance of these worms compared to reference site samples was much more important in contributing to dissimilarity between samples.

BEST analyses incorporating a range of factors potentially influencing benthic macrofaunal communities at the Central Works and Southern Works outfalls were performed. These included sediment characteristics, and Enrichment Factors for total organic content, chemical oxygen demand and a suite of trace metals. BEST analyses returned a highest Spearman correlation of ρ = 0.583 for a set of five sediment characteristics (very coarse-grained sand, medium-grained sand, very fine-grained sand,

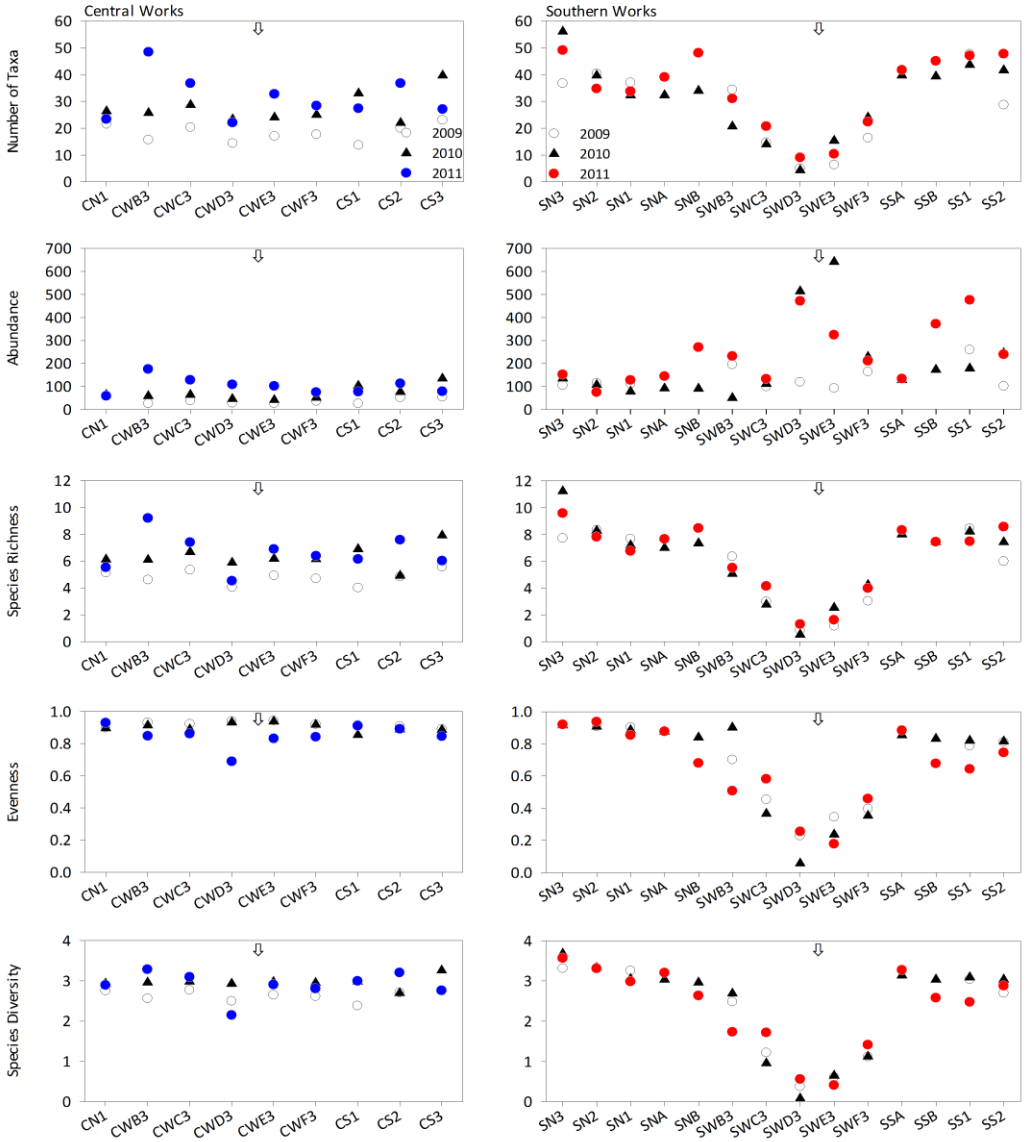

Figure 5.4. Indices of macrobenthic community structure for the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. Error bars are not included for presentation purposes. Arrows denote the positions of sites relative to the outfall diffusers.

and Enrichment Factors for copper and manganese) for Central Works outfall, suggesting these characteristics are the most influential (of those measured) in 'driving' biological variability. For the Southern Works outfall, a higher Spearman correlation of ρ = 0.602 was found using four characteristics (very fine-grained sand, total organic content, chemical oxygen demand, and cumulative metal Enrichment Factor).

5.3.3. 2011 Macrobenthic Survey Synopsis

The analyses discussed above reveal spatial trends in benthic macrofaunal communities similar to those noted in previous surveys, and particularly in recent surveys. The consistent findings for surveys performed between 2009 and 2011 provides a high level of confidence in the findings of the monitoring programme, and suggest that the behaviour of effluent in the receiving environment, and the response of the affected biota, has been consistent over recent years.

Significant differences exist between benthic macrofaunal communities in the immediate vicinity of the Southern Works outfall and communities at reference sites (≥1000 m from outfall). The differences manifest as reduction in number of taxa and abundance of pollution intolerant species, and marked increases in abundance of a few pollution tolerant species. This spatial trend in benthic macrofaunal community metrics follows closely the conceptual model proposed by Pearson and

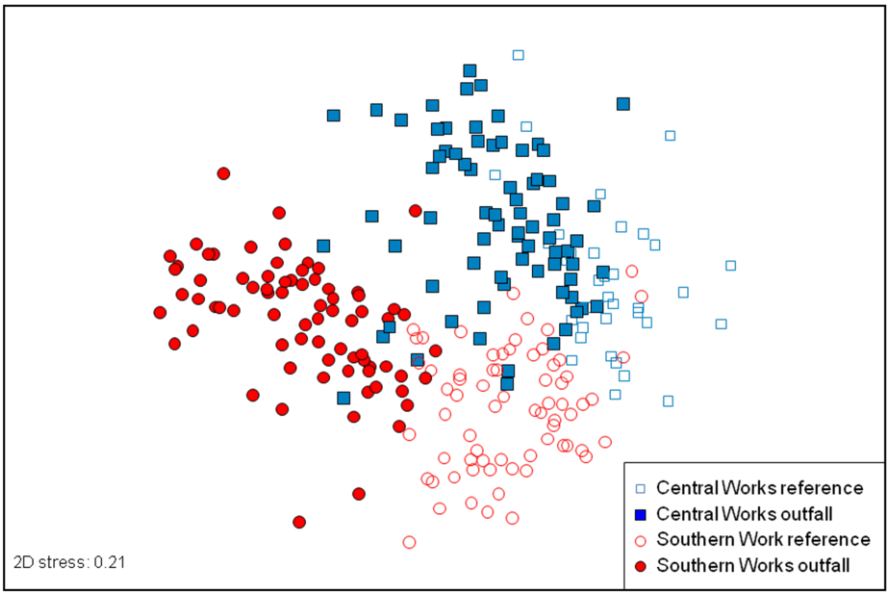

Figure 5.5. MDS ordination of macrobenthic samples taken from all outfall and reference stations in the 2009, 2010 and 2011 surveys.

Rosenberg (1978) for structural changes in soft-bottom benthic communities resulting from organic enrichment.

The superabundance of capitellid polychaetes (most notably those of the genus *Capitella*) at the Southern Works outfall site is particularly indicative. *Capitella* polychaetes are opportunists that thrive under organic enrichment. This is driven by several related factors. The first is the production of hydrogen sulphide in organically enriched sediments by bacteria under anaerobic conditions. Sulphides act as a settlement cue, attracting and promoting larval settlement of *Capitella* (Cuomo 1985). They are, however, toxic to a wide range of other species (Wang and Chapman 1999, and references therein). *Capitella*, therefore, recruit and successfully settle in areas of sulphide accumulation, which are sparsely populated by potential competitors due to their inability to tolerate the high sulphide concentrations. Once settled, these worms exploit a rich food supply. Indeed, there is a relationship between organic enrichment, growth rate and fecundity in capitellid polychaetes (Tsutsum 1990, Tsutsum et al 1990, Linton and Tagho 2000).

As in previous surveys for the Durban outfalls monitoring programme, potential toxicity effects on macrobenthos at the Southern Works outfall are difficult to determine. Sediment toxicity testing performed as part of this monitoring programme relies upon an extremely sensitive indicator, namely the fertilisation success of sea urchin gametes. BEST analysis indicated some correlation between benthic community patterns and concentrations of metals in sediment at both outfall sites, but the confounding effect of sediment grain size is difficult to separate. At both outfall sites, sediment granulometry appears to be the key driver of benthic macrofaunal community structure. Indicators of organic loading (total organic content, chemical oxygen demand) emerged as important additional factors in the case of the Southern Works outfall.

Benthic macrofaunal community differences between outfall and reference sites for the Central Works outfall were much less pronounced than those for the Southern Works outfall. This is reflected in low ANOSIM R values. Such differences have been noted in previous surveys, but they were not manifest in any of the univariate indices typically used in pollution studies and were not driven by differences in the abundance of capitellid polychaetes or any other classic pollution indicator. However, in 2011, preliminary indications are that a

Table 5.1. ANOSIM R values for pairwise comparisons of macrobenthic samples taken in the 2009, 2010 and 2011 surveys (p< 0.1 in all cases).

	CW outfall	CW ref	SW outfall
CW ref	0.226		
SW outfall	0.686	0.886	
SW ref	0.510	0.495	0.673

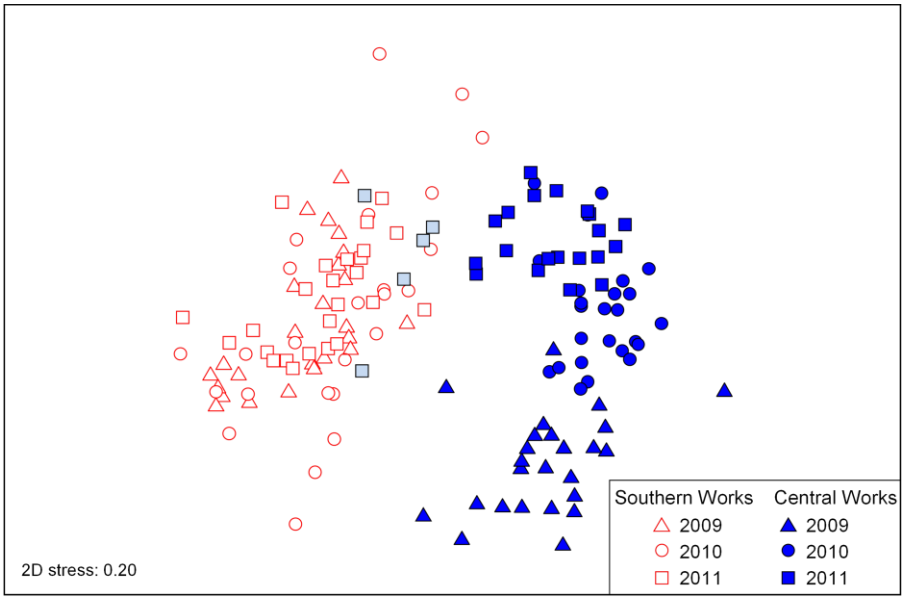

Figure 5.6. MDS ordination of macrobenthic samples taken from all outfall stations in the 2009, 2010 and 2011

relative abundance of capitellid polychaetes had a greater influence in driving community differences between Central Works outfall and references sites compared to surveys performed in 2009 and 2010. This is most likely a sporadic event that has been noted intermittently over the longer history of the outfalls monitoring programme, and will be monitored closely in future surveys. At present it is not regarded as a major concern.

Thus, in 2011 (as in 2009 and 2010), marked pollution effects were noted in benthic macrofaunal communities in close proximity to the Southern Works outfall. These impacts are restricted to an area <1000 m to the north-northeast of the outfall diffuser and a smaller distance to the south-southwest. Organic loading by sewage is the most likely cause of the impacts. In contrast, slight differences in benthic macrofauna were noted in the vicinity of the Central Works outfall. These were possibly the early results of impaired water or sediment quality in close proximity to the outfall diffuser, but this cannot be confidently asserted at this stage. Benthic macrofauna in close proximity to the Central Works outfall diffuser are still abundant, diverse and reflective of generally healthy conditions typical of the ocean shelf off the eThekwini municipal area.

5.3.4. Long-Term Trends

The sampling design for the Durban outfalls monitoring programme was most recently revised in 2009. A full analysis of the 2009, 2010 and 2011 combined dataset was performed for the purposes of this assessment. Spatial trends in univariate community indices tracked each other closely for the three surveys considered here, and reflect a consistent pattern of degradation in the immediate vicinity of the Southern Works outfall but otherwise generally healthy benthic macrofaunal communities elsewhere (Figure 5.4). As discussed above, in 2011, univariate indices reflected slightly skewed community structure at some stations near the Central Works outfall, potentially an early indication of impact from enrichment.

The output of multi-dimensional scaling on the full multivariate dataset is given in Figure 5.5. Although this ordination plot has a relatively large stress (0.21), groupings evident and discussed below were validated by CLUSTER analysis. Southern Works outfall samples in 2009, 2010 and 2011 clearly formed a distinctly separate cluster from the reference samples, and indeed all other samples. There is a less distinct (but still evident) separation of Central Works outfall and reference samples. In both cases, outfall samples separate from reference samples along a 'northwest-southeast' axis in Figure 5.5.

There is also a separation of Central Works from Southern works sites along a 'northeast-southwest' axis (Figure 5.5). This has been a persistent feature of this monitoring programme over the years and may be a depth related phenomenon, with the Southern Works outfall line of sites about 10 m

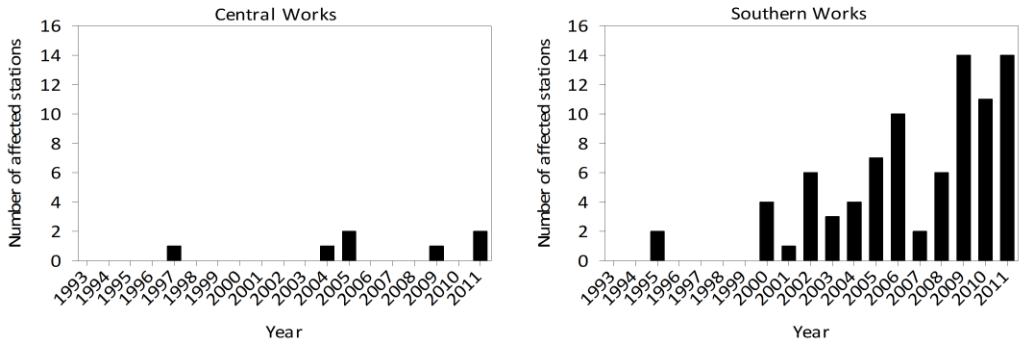

Figure 5.7. Trends in the number of 'affected' stations for the grids of stations spanning the diffuser sections of the Central Works and Southern Works outfalls between 1994 and 2011.

deeper than the Central Works outfall line of sites.

Pairwise comparison test results are given in Table 5.1. ANOSIM R values are an indication of the degree of discrimination between samples. Values close to unity (1) indicate complete discrimination while values close to zero (0) indicate inconsequential differences. From Table 5.1 it is evident that Southern Works outfall samples are most different to all other samples, while the difference between Central Works outfall reference and outfall samples is small (albeit statistically significant). SIMPER analysis indicated that the single taxon most responsible for driving dissimilarities between Southern Works outfall samples and all other groups of samples was capitellid polychaetes.

Figure 5.6 provides an analysis of the outfall samples only (i.e. omitting the reference samples) for surveys between 2009 and 2011. This provides more detailed insight into the response of the macrobenthos in close proximity to the Central Works outfall. Samples from the outfalls show almost complete separation in 2009 and 2010, as expected, with Southern Works outfall sites known to be impacted by organic enrichment from the effluent discharge and Central Works sites known to be unimpacted by effluent discharge. In 2011, however, potentially impacted Central Works outfall sites cluster with similarly impacted sites at the Southern Works outfall (lightly shaded squares in Figure 5.6). These samples correspond to those identified by univariate analysis as reflecting a slightly skewed community structure.

An analysis of longer term trends (1994 - present) can be performed using counts of 'affected' stations. The analysis of stations with evidence of impact ('affected' stations) is inevitably subjective and is based on composite conclusions drawn at the completion of each survey. It includes those stations where diversity and abundance were reduced (Shannon-Wiener diversity value <2) and/or numbers of capitellid polychaetes were increased to a level where they comprised more than 30% of the total count. It must be stressed that there are no universal criteria that can be applied in delineating impact. Each area has unique circumstances. The criteria adopted here are simply derived on empirical evidence and appear to make sense under current circumstances. In order to accommodate the most recent sampling design introduced in 2009, the analysis is restricted to the grid of stations/sites spanning the diffuser sections of the outfalls.

The number of 'affected' stations at the Central Works and Southern Works outfalls over the past eighteen years is provided in Figure 5.7. The difference between the outfalls is immediately apparent, with minor and infrequent impact noted at the Central Works outfall. In 2011, two stations in close proximity to the Central Works outfall were regarded as affected. These stations were also identified by other analyses as potentially reflecting benthic macrofaunal communities influenced by organic enrichment of sediment. This is presently of little concern, but will be closely monitored in future surveys.

In contrast, there has been a persistent trend toward decreased diversity in the vicinity of the Southern Works outfall, with the number of 'affected' stations increasing significantly since 2000 (Figure 5.7). The number of 'affected' stations in 2011 has increased to the same highest level recorded in 2009, with 14 of the 15 stations regarded as being affected. It is important to bear in mind the relatively small spatial

scale of concern in this context - the stations are all within 500 m of the outfall diffuser. The changes that are currently manifesting over this spatial scale are considered to be no cause for immediate concern. They would be expected near a large outfall and, in a sense, provide reassurance of the sensitivity of the monitoring methodology. However, the increase in the number of affected stations over the last decade is cause for concern in the long-term and must remain a factor in management considerations.

5.4. Conclusions

Univariate and multivariate analysis of benthic macrofaunal community structure provide clear evidence that the seabed near the Southern Works outfall is enriched with particulate organic material. Benthic macrofaunal community structure in close proximity to the outfall has been modified because of this enrichment. This is manifested by reduced biodiversity and an increased abundance of opportunist capitellid polychaetes. Comparison with earlier surveys reveals a gradual increase of this effect over the past decade. While this impact is not considered to pose an immediate ecological threat, its expansion is cause for concern and should be accounted for in management considerations, especially if effluent volumes were to increase.

The Central Works outfall appears to be operating within the assimilative capacity of the receiving environment. In 2011, however, some indication was given of impacts manifesting in benthic macrofauna in close proximity to the outfall. Benthic macrofaunal community response at affected stations appears to be similar to that at impacted stations near the Southern Works outfall and is likely the result of mild organic enrichment of the sediment. This is presently of little concern but requires close monitoring in future surveys.

5.5. References

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5.6. Appendices

Appendix 5.6.1. Macrobenthos taken near the Central Works outfall and at reference sites for the 2011 survey of the Durban outfalls monitoring programme.

Taxa	CN1A	CN1B	CN1C	CWB2	CWB3A	CWB3B	CWB3C	CWB4	CWC2	CWC3A	CWC3B	CWC3C	CWC4	CWD2	CWD3A	CWD3B	CWD3C	CWD4	CWE2
Acharax spp						4	1					1	6	1			1	1	1
Actiniaria spp				1															
Albunea symnista												1					1		
Amaryllis macrophthalma														1					
Ampelisca brevicornis	2		2	4					4	2							2		2
Ampelisca diadema							6												1
Ampelisca miops																			
Ampelisca natalensis			2	3	10		5			2		6	2	1		1			
Ampelisca palmata						6													1
Ampelisca spinimana							12			3	4								
Ampelisca spp															1		2	2	1
Ampharetidae spp						1	1	1				2							
Amphilochidae spp								1											
Ancistrosyllis parva					1														
Anomura spp					2	1	7	6			2	1		1		3		1	1
Anthuridae spp		1																	
Aonides oxycephala				1	1	3		23				1	2		2	3		6	
Arabella iricolor					1				1	2		3	3				1		
Arcturina spp							1												
Armandia leptocirrus										1									
Armandia spp				1															
Astropecten spp		1	1																
Bullia similis		2																	
Byblis gaimardi			1		4	4	1	1		2			1						
Capitella spp					1	74		25		2	1	5	27	8	52	71	2	57	
Capitellidae spp	1	1	4		4	1	3		2									2	
Caprellidae spp																			
Caridea spp		4			5	2	3	5	6		7	2		4		5	1	5	4
Chaetopteridae spp					1	1	2	2	1	2		1	3	2				1	
Chevalia aviculae					1														
Cirolana spp																			
Cirratulidae spp					3		1				1	2		1			1	2	2
Corophiidae spp			1		11	6	3	3	3			3				7	2	7	
Cunicus profundus																			
Cyclaspis australora																			
Dehaanius spp																		1	

Taxa	CN1A	CN1B	CN1C	CWB2	CWB3A	CWB3B	CWB3C	CWB4	CWC2	CWC3A	CWC3B	CWC3C	CWC4	CWD2	CWD3A	CWD3B	CWD3C	CWD4	CWE2
Dexaminidae spp																			
Dic calmani																			
Dic formosae				1															
Diopatra cuprea						1			1							2			
Diopatra neopolitana					1	4			2	1		2				1			1
Diopatra spp				1				1											
Dispio magnus								2											
Donax spp									4										1
Dorvillea spp																3			
Echinoidea spp					1														1
Echiurida spp									1										
Eocuma winri			2		1			1					1						
Epidiopatra spp																			
Eteone sp																			
Eunice spp							2	1											
Euthalenessa oculata																			
Flabelligeridae spp			2							1									
Galathea spp																2		1	
Gammaropsis spp				1						1		3	2						
Glycera spp	2	1	1	1	4	14	7	8	2	1	2	5	5	6	3	1		9	4
Glycinde capensis	3		3	3	7	3	9	10	4	6	12	8	6	4	3	9	3	6	13
Gnathia spp																			
Golfingia spp			1							1									
Gynodiastylis curvirostris													2						
Haminoea spp					4		1												
Hippomedon normalis	6	5	10	18	1		6		13	15	4		1	4			11		25
Hippomedon onconotus										12									
Inachidae spp							1					1				1			
Iphinoe crassipes	2	5	3	5		1			1		1		1	1			4	1	1
Iphinoe stebbingi																			
Laetmatophilus spp																1			
Leda gemmulata			1		1														
Leptochela spp		1	2	2	2	3	2	1	4	3	4	1		11					8
Leucothoidae																1			
Liljeborgiidae spp				1														1	
Lucifer spp											1						1		
Lumbrineris spp		1				3				3			1						
Lysianassa spp	1																		
Lysianassidae							1												
Macoma spp					1		1					1	1			2			

Taxa	CN1A	CN1B	CN1C	CWB2	CWB3A	CWB3B	CWB3C	CWB4	CWC2	CWC3A	CWC3B	CWC3C	CWC4	CWD2	CWD3A	CWD3B	CWD3C	CWD4	CWE2
Magelona cincta						1	1							2					
Mandibulophoxus stimpsoni	5	1	10	12	4	1	2		13	13		8	1	1			6		11
Mysidacea spp					2	1	1	1		1				3		2	2		1
Natica gualteriana								1						1				1	
Neastacilla sp																	1		
Nebalia capensis				1									1						
Nemertea spp		1			1							1							
Nephasoma rutilofusca								1	1										
Nephtys dibranchus					2	4	5				4		3						
Nephtys hombergi	2		2	2				1	5	2	1	2		2	1		2	1	2
Nephtys sp	2																		
Nereis sp						1										8		11	
Nerinides spp																			
Notomastus spp						1		1					1					2	
Nudibranchia spp									1										
Octocorallia spp					1			1	3		1					4		5	
Oedicerotidae spp				1	2		1					1							1
Oliva spp																			
Onuphis conchylega													2					1	
Onuphis eremita	6		4	11	4	6	2		7	4	1	9	1	7	1			3	5
Onuphis spp			1	1		1													
Ophelia spp																			
Ophiuroidea spp					1		2	1	1	1	2	2				6	1		4
Orbinia bioreti						1													
Orbinia cuvieri			3	1		1				1			2						2
Orbiniidae spp																1			
Ostracoda spp	2	0	3	10	11	21	12	7	37	41	16	14	7	6	3	22	14	8	12
Owenia fusiformis				3										2					
Paguridae spp			1																
Paraonidae spp		2						1											
Pareulepis geayi						1			1										
Pectinaria spp					3		3	1				1	1						
Pennatulacea spp									1										
Petricola sp						3		1		3	1	1	4						
Petrolisthes sp						1													
Phaxas sp											1								
Philine aperta						3						5							
Philyra spp					2		8	4			5			2		1			
Pholoe sp					3		4	1			1								

Taxa	CN1A	CN1B	CN1C	CWB2	CWB3A	CWB3B	CWB3C	CWB4	CWC2	CWC3A	CWC3B	CWC3C	CWC4	CWD2	CWD3A	CWD3B	CWD3C	CWD4	CWE2
Photis spp	1	2	2		2	1	5	3						2				2	
Phyllodoce madeirensis			1		1				1									1	
Phyllodoce malmgreni							1												
Phyllodoce spp														1					
Phylo spp				1		3							2						
Pinnotheres sp							1				1			1		1			
Platyischnopus herdmani	1	4	2	3					4								6		1
Podocerus spp									1										2
Poecilochaetus serpens							1						1					3	
Polydora spp		1				1	1		1	2				1					2
Polyodontes sp																			
Portumnus spp					1														
Portunidae spp										1									
Portunus (Monomia) gladiator																			
Prionospio ehlersi																			
Prionospio malmgreni																			
Prionospio pinnata				3	1			1	4		1	1	2	1					2
Prionospio saldanha	2	1		1		4		2				1	1		1				
Prionospio spp		2	3	2	5		4		3	6			4	2		5	2	2	3
Prionospio steenstrupi																			
Sabellidae spp			2	5	1	2	3		4	5	3	3		5		2	3		
Schizammina pinnata			1	1	1				1	1		1		1					
Scoloplos spp					1	1		2				1	1						
Sigalion capense																			
Siliqua fasciata						1													
Sipuncula spp												1			1				1
Solenocera sp							1												
Spio spp					2	1				2			1						
Spiochaetopterus spp				2	14	4	4	3	2	5	4	9	2	2			1		
Spionidae sp				1						1	1		1	1	2				
Spiophanes bombyx				1	2		1		2	3		3		2			2	1	2
Spiophanes soederstromi		1		1	1			1											
Talitridae spp					2	1			1	1	1	1	1	2			3		2
Tellina spp						1		1	1		1	1	2						
Tellina vidalensis				1	2				2				1	5					1
Terebellidae spp					4							5		1					
Tharyx spp			1	2				2			2								
Timoclea arakana	4	2	3	9	17	8	9	1	9	7	1	10	5	9	1		1		
Turbellaria spp																			

Taxa	CN1A	CN1B	CN1C	CWB2	CWB3A	CWB3B	CWB3C	CWB4	CWC2	CWC3A	CWC3B	CWC3C	CWC4	CWD2	CWD3A	CWD3B	CWD3C	CWD4	CWE2
Unciolella spp						4													
Urothoe elegans	2		4						4								4		
Urothoe pinnata																			
Urothoe pulchella		1	10	6	1	1			3	2		2	1	2			6		11
Urothoe spp							1	1											
Xanthidae spp						1													

Appendix 5.6.1 continued. Macrobenthos taken near the Central Works outfall and at reference sites for the 2011 survey of the Durban outfalls monitoring programme.

Taxa	CWE3A	CWE3B	CWE3C	CWE4	CWF2	CWF3A	CWF3B	CWF3C	CWF4	CS1A	CS1B	CS1C	CS2A	CS2B	CS2C	CS3A	CS3B	CS3C
Acharax spp		1	1	6				1						1				
Actiniaria spp																		
Albunea symnista											1				1			
Amaryllis macrophthalma															1		6	1
Ampelisca brevicornis	2	2			5	1			1	1		1	1	1	4			
Ampelisca diadema	1	2	4	1		2												
Ampelisca miops			6										2			1	2	1
Ampelisca natalensis							3		2					1				
Ampelisca palmata			4															
Ampelisca spinimana																		
Ampelisca spp																		
Ampharetidae spp																		
Amphilochidae spp								1										
Ancistrosyllis parva																		
Anomura spp		1	2		1	1			1			1		1				1
Anthuridae spp				1		1												
Aonides oxycephala				2			1											
Arabella iricolor	1						1											
Arcturina spp																		
Armandia leptocirrus					3		1	2					1	1			1	
Armandia spp										1								
Astropecten spp																		
Bullia similis					2					1	1	2						
Byblis gaimardi					1				1	1								
Capitella spp	3	2	3	20		1												
Capitellidae spp					1		2					2				1	3	
Caprellidae spp			1															
Caridea spp	2		3		4	5		1		1	3		1	5	1			
Chaetopteridae spp		2				1			1				1					

Taxa	CWE3A	CWE3B	CWE3C	CWE4	CWF2	CWF3A	CWF3B	CWF3C	CWF4	CS1A	CS1B	CS1C	CS2A	CS2B	CS2C	CS3A	CS3B	CS3C
Chevalia aviculae																		
Cirolana spp											2					2		2
Cirratulidae spp		1		1						1	1	1						1
Corophiidae spp		1	2			1									2		3	
Cunicus profundus											1	1						3
Cyclaspis australora																		2
Dehaanius spp			1															
Dexaminidae spp											1							
Dic calmani												1						2
Dic formosae																		
Diopatra cuprea															1			
Diopatra neopolitana																		
Diopatra spp														2			1	
Dispio magnus																		
Donax spp					1					1	1	2		2				
Dorvillea spp																	1	
Echinoidea spp			1															
Echiurida spp																		
Eocuma winri	1			2	3	1				1			1					
Epidiopatra spp			1					1										
Eteone sp						1					1				1			
Eunice spp							1									1		
Euthalenessa oculata																		2
Flabelligeridae spp								2										
Galathea spp																		
Gammaropsis spp					1													
Glycera spp	3	2	4	7	2	2	5	2	3	1		1	4	6	2			1
Glycinde capensis	10	3	2	4	9	9	3	4	7	5	5	2	4	5	5			1
Gnathia spp																1		
Golfingia spp													1	1				
Gynodiastylis curvirostris																		
Haminoea spp																		
Hippomedon normalis	4	6		6	36	6	10	1	5	11	3	4	5	9	12	5	8	5
Hippomedon onconotus														7				
Inachidae spp																		
Iphinoe crassipes					22	1		2	2	15	2	1	6	1	11			1
Iphinoe stebbingi														2				
Laetmatophilus spp																		
Leda gemmulata													1	3				
Leptochela spp				4	1	2		2		3	3		1		3		1	

Taxa	CWE3A	CWE3B	CWE3C	CWE4	CWF2	CWF3A	CWF3B	CWF3C	CWF4	CS1A	CS1B	CS1C	CS2A	CS2B	CS2C	CS3A	CS3B	CS3C
Leucothoidae				123											1			
Liljeborgiidae spp																		
Lucifer spp		2				1							1					
Lumbrineris spp	5			2		1	2		1						1		1	
Lysianassa spp	1			1			1											
Lysianassidae													2					
Macoma spp							1					1						
Magelona cincta				1														
Mandibulophoxus stimpsoni	1	4	1		30	2	2	3	1	4	2	5	9	13	6	11	10	25
Mysidacea spp	1				5					3	1			1	1			1
Natica gualteriana										1			1					
Neastacilla sp																		
Nebalia capensis																4		2
Nemertea spp										1			1					
Nephasoma rutilofusca		1																
Nephtys dibranchus			6	2														
Nephtys hombergi	2		2						3					2	1		8	3
Nephtys sp	1							2			3		1	2	1	1		1
Nereis sp			1												1			
Nerinides spp	1															1		
Notomastus spp		1																
Nudibranchia spp													1					
Octocorallia spp				4														
Oedicerotidae spp	1										3	1	3	2				1
Oliva spp																1		
Onuphis conchylega	1			2														
Onuphis eremita	3	3	9	1	3	6	2	3	1	1	1			1	3			5
Onuphis spp							1											
Ophelia spp																1		1
Ophiuroidea spp		2	1	2	1					1	1			1		1		
Orbinia bioreti																		
Orbinia cuvieri	1				2	2	1		2	5			1	2	3	1	3	1
Orbiniidae spp																		
Ostracoda spp	12	54	17	4	18	35	15	12	14	2	1	5	3	1	2	2	0	3
Owenia fusiformis															1			
Paguridae spp																	1	
Paraonidae spp					1													
Pareulepis geayi						1		1	1									
Pectinaria spp			4	2														

Taxa	CWE3A	CWE3B	CWE3C	CWE4	CWF2	CWF3A	CWF3B	CWF3C	CWF4	CS1A	CS1B	CS1C	CS2A	CS2B	CS2C	CS3A	CS3B	CS3C
Pennatulacea spp																		
Petricola sp	1		1	5					4									
Petrolisthes sp																		
Phaxas sp																		
Philine aperta																		
Philyra spp			3		1													
Pholoe sp	1	3			2		1		2									
Photis spp			1		1				1	1			1	4				
Phyllodoce madeirensis			2					1										
Phyllodoce malmgreni																		
Phyllodoce spp		2		4							1			1	1		1	
Phylo spp																	1	
Pinnotheres sp																1		
Platyischnopus herdmani	3				5	1		2	2	11	4	7		12	5			
Podocerus spp					1					1					1			1
Poecilochaetus serpens				2														2
Polydora spp						1							1	1	2			1
Polyodontes sp						1												
Portumnus spp																		3
Portunidae spp				1											1			
Portunus (Monomia) gladiator										1								
Prionospio ehlersi															1			
Prionospio malmgreni					2	1	1		3									
Prionospio pinnata	1	1				3	1	3						1	4	1		1
Prionospio saldanha		1						1					3	3			3	
Prionospio spp	6	2	5	1			1	2		5	5	2	9	10	8	6	1	7
Prionospio steenstrupi															1			
Sabellidae spp	2		1		2			1	1	1		1	1	2	5			
Schizammina pinnata										1						1	1	1
Scoloplos spp					1										1			
Sigalion capense		1													2	1		1
Siliqua fasciata																		
Sipuncula spp																		
Solenocera sp																		
Spio spp			1											1				
Spiochaetopterus spp	5	3	2	4		1	2			2								
Spionidae sp		1									2	3			4			
Spiophanes bombyx					1								1		2	1		1
Spiophanes soederstromi	1	1	1	1			1	1							1			

Taxa	CWE3A	CWE3B	CWE3C	CWE4	CWF2	CWF3A	CWF3B	CWF3C	CWF4	CS1A	CS1B	CS1C	CS2A	CS2B	CS2C	CS3A	CS3B	CS3C
Talitridae spp	2	1			1	1		1		1								
Tellina spp	2			3	2				3									
Tellina vidalensis		1	4			2	3											1
Terebellidae spp							1						1					
Tharyx spp			1	1	2					3	4	2						
Timoclea arakana	4	4	4	2	2	1	1	1							1	1	2	
Turbellaria spp					1													
Unciolella spp																	2	
Urothoe elegans	3				1					12		4		4	2		1	3
Urothoe pinnata					4					10			10	6	7	3		
Urothoe pulchella					11	2	1	6		10	5			13	12		3	32
Urothoe spp																		
Xanthidae spp			1															

Appendix 5.6.2. Macrobenthos taken near the Southern Works outfall and at reference sites for the 2011 survey of the Durban outfalls monitoring programme.

Taxa	SN3A	SN3B	SN3C	SN2A	SN2B	SN2C	SN1A	SN1B	SN1C	SNAA	SNAB	SNAC	SNBA	SNBB	SNBC	SWB2	SWB3A	SWB3B	SWB3C
Acharax spp		1																	
Actiniaria spp	2				1	2													
Albunea symnista																			
Amaryllis macrophthalma		2																	
Ampelisca brevicornis											1		1				2		
Ampelisca diadema	1		3	1	2		2		6		2	3		5			1		
Ampelisca miops	7		3	5	1	2	6			6	3	4		5	4				
Ampelisca natalensis							3			2	4								
Ampelisca palmata								2	3	5			2		5		6		
Ampelisca spinimana									5				4	4	3		9		9
Ampelisca spp				1				1										1	1
Ampharetidae spp							1						1		1				
Ancistrosyllis parva		3	1							1		1							
Anomura spp	2	2			1	4	53		1			1		2		1	2		
Anthuridae spp		9	8				1												
Aonides oxycephala								1	2			2	2		3	8	10	11	18
Apionosoma trichocephala	2	2	4																
Arabella iricolor	1	1	1		1														
Arcturinoides sexpes				1								1							
Armandia spp																			
Ascidiacea spp		1																	
Branchiomma nigromaculatum	3	2	2		1														
Bullia similis																			
Byblis gaimardi	1	1	1	1	1		4		2	5		2		1		1	4	2	
Capitella spp		1					19	19	18	17		15	32	172	153	7	207	129	103
Capitellidae spp	4		3	1	4	3					2								
Caridea spp	1	1		3		1	5	1	1				2	2	1		5	2	2
Chaetopteridae spp	2	5	4	3			1	2		1	1		2	4	4			1	
Charybdis sp																			
Chevalia aviculae													1		2		1		
Chlamys sp	1	1																	
Cirolana spp														1					
Cirratulidae spp	1		2			2			1	1		1			1	1			
Corophiidae spp	13	2	7	7	5	5	4	3	1	6	2	21	25	12	8	1	4	4	12
Cunicus profundus																			
Cyclaspis scissa			1																
Dehaanius spp							1												

Taxa	SN3A	SN3B	SN3C	SN2A	SN2B	SN2C	SN1A	SN1B	SN1C	SNAA	SNAB	SNAC	SNBA	SNBB	SNBC	SWB2	SWB3A	SWB3B	SWB3C
Diopatra cuprea	10	3		4	7	2	1		1	1		1	5	2			1		2
Diopatra neopolitana												1							1
Diopatra spp																			
Dispio magnus																			
Donax spp										1					1				
Dorvillea spp	1							2				1		1	1				
Dromiidae spp															1				
Ebalia spp													2	1					
Echinocardium spp		1					2							1					
Echinoidea spp																			
Eocuma winri							1												
Epidiopatra spp																	1		
Eunice spp	5	5		1	6	2	1	1	3	3	1	1	3	1	5	3	2		1
Eusiridae spp																			
Flabelligeridae spp		3													1				
Galathea spp	1	5	2			2				2			2	2		1	4		2
Gammaridae spp		1														1			
Gammaropsis afra	5		6	1															
Gammaropsis spp	8	3			4		3					5	8	9	3	1	4		2
Glycera spp	6	1	2	4	2		6	4	2	5	2	7	2	5	8	5	15		3
Glycinde capensis	8	2	11	5	5	8	5	3	11	11	6	10	8	6	14	6	3	1	2
Golfingia spp	4	2	2		1	4													
Gynodiastylis curvirostris									1										
Haminoea spp													1						
Harmothoe spp	3	3	1			1				1	1	1	1		1				
Hesionidae spp												1							
Hippomedon normalis				2						9				7	3				
Hippomedon onconotus								8	6	10	16	20	15	11	3	2			
Inachidae spp		2																	
Iphinoe crassipes										2	1								
Lanocira gardineri																			
Lasaeidae spp	2	1			1														
Leptochela spp	1	1				2	2	2				2	2	2	1			1	2
Leucothoidae	9									1									
Lumbrineris spp	1		1								1		1		1			2	
Lysianassa spp							3												
Lysianassidae																			
Macoma spp																			
Magelona cincta					1		1												
Maldanidae spp	8	4	4	5	2	1													

Taxa	SN3A	SN3B	SN3C	SN2A	SN2B	SN2C	SN1A	SN1B	SN1C	SNAA	SNAB	SNAC	SNBA	SNBB	SNBC	SWB2	SWB3A	SWB3B	SWB3C
Mandibulophoxus stimpsoni										7	4	2	6	3			1		
Mysidacea spp									1										
Nassarius sp						1		1											
Natica gualteriana																			
Nebalia capensis																			
Nemertea spp			1															1	
Nephasoma rutilofusca						1													
Nephtys dibranchus				2	2														
Nephtys hombergi	2		1			3		4	2	6	7	2	2	1	2		2		2
Nephtys sp	1	2		1															
Nereis sp	11	5		2		1				2	1		3	2	1		3	1	4
Nerinides spp															1		1		
Notomastus spp			1					1				1	1		1		3		
Notophyllum splendens																			
Octocorallia spp																	1		
Oedicerotidae spp				1		2			1				1		1		1		
Onuphis conchylega																			
Onuphis eremita	3	2	1	5	3	1	3	5	3	1	3	3	7		1	2	6	5	3
Onuphis spp				1															
Ophiuroidea spp	16	7			1		1							2		1			
Orbinia bioreti				1	2													1	
Orbinia cuvieri				2	1					4	3	3	2	4	2	2	1		
Orbinia spp		1																	
Orbiniidae spp					1					1									
Ostracoda spp	3	4	0	5	2	3	0	7	3	9	3	8	3	4	1	0	0	1	1
Owenia fusiformis	1																		
Paguridae spp									1										
Paraonidae spp										2		1							
Parvicardium turtoni	1	2																	
Pectinaria spp																			
Petricola sp																			
Philine aperta										1									
Philyra globulosa																	1		
Philyra spp				4			3		1					3	3		2		2
Pholoe sp					1							2	1	2					
Photis spp	2	2	4	1			6	3		3		2	2	6	2	1	7		
Phyllodoce madeirensis	1	2		1		1	2	2					1	1	1			1	
Phyllodoce spp	3	1			2		9							1	2		1		
Phylo spp							1						1				1	3	1

Taxa	SN3A	SN3B	SN3C	SN2A	SN2B	SN2C	SN1A	SN1B	SN1C	SNAA	SNAB	SNAC	SNBA	SNBB	SNBC	SWB2	SWB3A	SWB3B	SWB3C
Pinnotheres sp															1				
Platyischnopus herdmani	1	1		1							1	1	1			1			
Platylambrus quemvis													1						1
Podocerus spp																			
Poecilochaetus serpens					1		2	1			1	1		1	1		1	1	
Polydora spp			1				1				1								
Porcellana spp	9	16	1		1		3						3				2		
Portunidae spp	1																		
Prionospio ehlersi																			
Prionospio pinnata																			
Prionospio saldanha				1	1				2	1	1		1		2	1			
Prionospio spp	3	4	7	2	1	1		12	5	8	5	5	2	2	8	2	4	3	2
Pseudojanira spp	1																		
Pycnogonida sp								1	2										
Sabellidae spp			2					2	4	2	1	2	1	13	5		3	1	2
Schizammina pinnata		1																	
Scoloplos spp							2	4	2										
Semele spp										1									1
Serpulidae spp													1						
Sigalion capense	2																		
Sipuncula spp																			
Spio spp	1	1	2	4			1			1		3	1	1	2				1
Spiochaetopterus vitrarius	5	3	8	2	2	2		3	5	3	2	4	1	4	4	4	2	2	1
Spionidae sp													1						
Spiophanes bombyx	2									2	1		2						
Spiophanes soederstromi		4	3	1					2			2	2	1	2		1	1	
Sthenelais spp		3																	
Syllidae spp	1		2	1			3		2					1					
Talitridae spp										1				1	1		1	1	2
Tanaidacea spp	1																		
Tellina spp							2							1					
Tellina vidalensis					1					2	1								
Terebellidae spp	4	5	2	1		1				1			5						
Tharyx spp			1												2				
Timoclea arakana	2		1	1	2	1	4	1		1	3	2	2	1	2	1		1	
Unciolella spp		2		2	1		1					3	3	22	2		4	1	
Urothoe elegans	1	2	3	2			1	5	7	9	18	3	14	2	3				
Urothoe pinnata																			
Urothoe pulchella		2	1	1			1			5	9	7				2		1	
Xanthidae spp	4	4		1	1		1			1									

Appendix 5.6.2 continued. Macrobenthos taken near the Southern Works outfall and at reference sites for the 2011 survey of the Durban outfalls monitoring programme.

Taxa	SWB4	SWC2	SWC3A	SWC3B	SWC3C	SWC4	SWD2	SWD3A	SWD3B	SWD3C	SWD4	SWE2	SWE3A	SWE3B	SWE3C	SWE4	SWF2	SWF3A	SWF3B
Acharax spp																			
Actiniaria spp																			
Albunea symnista																		1	
Amaryllis macrophthalma												1							
Ampelisca brevicornis	1		1															1	
Ampelisca diadema	2										5								1
Ampelisca miops																5			
Ampelisca natalensis											1								
Ampelisca palmata						1													
Ampelisca spinimana	8		5		1	5					10	1				5		2	
Ampelisca spp	1																2	2	
Ampharetidae spp	1																		
Ancistrosyllis parva																			
Anomura spp			1			1					1				1	2	1		
Anthuridae spp	1																1		
Aonides oxycephala	4	2	10	5	14	7	9	1			7	12				5	10	14	6
Apionosoma trichocephala																			
Arabella iricolor																1			
Arcturinoides sexpes																			
Armandia spp																			
Ascidiacea spp																			
Branchiomma nigromaculatum																			
Bullia similis						2													
Byblis gaimardi																1	2		
Capitella spp	73	488	134	18	105	121	177	476	349	361	149	264	243	329	321	87	132	130	126
Capitellidae spp					1												3	1	
Caridea spp	4				3	2		2		2								1	
Chaetopteridae spp											1							1	
Charybdis sp	3																		
Chevalia aviculae																			
Chlamys sp																			
Cirolana spp																	1		
Cirratulidae spp																			
Corophiidae spp	11					4											4	3	
Cunicus profundus																			
Cyclaspis scissa																			

Taxa	SWB4	SWC2	SWC3A	SWC3B	SWC3C	SWC4	SWD2	SWD3A	SWD3B	SWD3C	SWD4	SWE2	SWE3A	SWE3B	SWE3C	SWE4	SWF2	SWF3A	SWF3B
Dehaanius spp																			
Diopatra cuprea	11	1			5	4		1			1		1					4	
Diopatra neopolitana					2														
Diopatra spp																1			
Dispio magnus													1						1
Donax spp	1																		
Dorvillea spp																	1		
Dromiidae spp																			
Ebalia spp																			
Echinocardium spp																			
Echinoidea spp																			
Eocuma winri																			
Epidiopatra spp																			
Eunice spp						1											2		
Eusiridae spp																		1	
Flabelligeridae spp																			
Galathea spp	7		1		1	3													
Gammaridae spp																			
Gammaropsis afra																			
Gammaropsis spp	8					2												2	
Glycera spp	6	2	7	3		9	5		4	1	9	6	5	7	6	10	16	16	14
Glycinde capensis	9	1	4	3		6	3		3	1	3	3	2	2	1	3	9	5	2
Golfingia spp																			
Gynodiastylis curvirostris																			
Haminoea spp																			
Harmothoe spp	1																		
Hesionidae spp																			
Hippomedon normalis																			
Hippomedon onconotus																			
Inachidae spp																			
Iphinoe crassipes																			
Lanocira gardineri																			
Lasaeidae spp																			
Leptochela spp	10		5	4	4	4	1			3	2		1			2	2	1	
Leucothoidae																			
Lumbrineris spp											1								
Lysianassa spp																			
Lysianassidae																			
Macoma spp																			
Magelona cincta					1														

Taxa	SWB4	SWC2	SWC3A	SWC3B	SWC3C	SWC4	SWD2	SWD3A	SWD3B	SWD3C	SWD4	SWE2	SWE3A	SWE3B	SWE3C	SWE4	SWF2	SWF3A	SWF3B
Maldanidae spp																	1		
Mandibulophoxus stimpsoni																			
Mysidacea spp	1		1		2	2						1			1				
Nassarius sp									1										
Natica gualteriana	2																		
Nebalia capensis																			
Nemertea spp										1							2		
Nephasoma rutilofusca																			
Nephtys dibranchus																			
Nephtys hombergi			1	1			1	2	1			3	2		1	1	3	1	2
Nephtys sp																			
Nereis sp	5	11		1		8		56	63	73	3		1	18	14		2	1	2
Nerinides spp											1					2			
Notomastus spp												1	1	1		1	1		
Notophyllum splendens																			
Octocorallia spp				2							1							1	
Oedicerotidae spp		1																	
Onuphis conchylega	4					1										3			
Onuphis eremita	5		5	1	4	3					3			1		2	3	2	4
Onuphis spp		1					1		1			4	1						4
Ophiuroidea spp	1				1														
Orbinia bioreti					1		1			1			1				1	1	
Orbinia cuvieri	2						1												
Orbinia spp				1															
Orbiniidae spp					1														
Ostracoda spp	1	0	0	0	0	1	0	0	0	0	0	2	0	0	1	0	3	0	1
Owenia fusiformis																			
Paguridae spp																			
Paraonidae spp																			
Parvicardium turtoni																	1		
Pectinaria spp																			
Petricola sp																			
Philine aperta	1																		
Philyra globulosa	1				1														
Philyra spp					1						1								
Pholoe sp																			
Photis spp			1	1															
Phyllodoce madeirensis							1												
Phyllodoce spp	2			1		1													

Taxa	SWB4	SWC2	SWC3A	SWC3B	SWC3C	SWC4	SWD2	SWD3A	SWD3B	SWD3C	SWD4	SWE2	SWE3A	SWE3B	SWE3C	SWE4	SWF2	SWF3A	SWF3B
Phylo spp	2	1	1	1	1	1			1		2					1	5	3	2
Pinnotheres sp																			
Platyischnopus herdmani																	1		
Platylambrus quemvis					1														
Podocerus spp																		1	
Poecilochaetus serpens					1	1		2	2	1	2	1		1			1	3	2
Polydora spp	1				1												1	1	
Porcellana spp	2																		
Portunidae spp				1		1													
Prionospio ehlersi														2					
Prionospio pinnata					1														
Prionospio saldanha					3														
Prionospio spp	3		2	1	3		2		2	1	2	1	1		1	2	6	9	7
Pseudojanira spp																			
Pycnogonida sp																			
Sabellidae spp	4		2			2					2						3		
Schizammina pinnata																			
Scoloplos spp																			
Semele spp																			
Serpulidae spp																			
Sigalion capense																			
Sipuncula spp																1			
Spio spp	1										2					1			
Spiochaetopterus vitrarius	2		1			4					3					1	3	3	2
Spionidae sp					3														
Spiophanes bombyx						1											1		
Spiophanes soederstromi						3												1	
Sthenelais spp																			
Syllidae spp															1				
Talitridae spp			2								2					1			
Tanaidacea spp																			
Tellina spp																			
Tellina vidalensis													1						
Terebellidae spp	1				1	2											2	2	
Tharyx spp																			
Timoclea arakana		3	1			1						1							
Unciolella spp	4		3		1												10		
Urothoe elegans																			
Urothoe pinnata																			
Urothoe pulchella	5																		

Taxa	SWB4	SWC2	SWC3A	SWC3B	SWC3C	SWC4	SWD2	SWD3A	SWD3B	SWD3C	SWD4	SWE2	SWE3A	SWE3B	SWE3C	SWE4	SWF2	SWF3A	SWF3B
Xanthidae spp																			

Appendix 5.6.2 continued. Macrobenthos taken near the Southern Works outfall and at reference sites for the 2011 survey of the Durban outfalls monitoring programme.

Taxa	SWF3C	SWF4	SSAA	SSAB	SSAC	SSBA	SSBB	SSBC	SS1A	SS1B	SS1C	SS2A	SS2B	SS2C
Acharax spp			1	1	1	2	2							
Actiniaria spp			2									1	1	
Albunea symnista						1								
Amaryllis macrophthalma											1	4		9
Ampelisca brevicornis		1	2	3		6	1	6	15	4	9	8	5	8
Ampelisca diadema		1	1					3						
Ampelisca miops		3				3			8		1		4	
Ampelisca natalensis														
Ampelisca palmata						7	1			5			3	
Ampelisca spinimana		5	4	4		8	8	3		18	4	6	7	5
Ampelisca spp	1	2												1
Ampharetidae spp						1								
Ancistrosyllis parva			1		1				1					
Anomura spp		2		1	2	5		3		2				
Anthuridae spp					1	1	2	6		4	2	6		2
Aonides oxycephala	6	9	2		3									
Apionosoma trichocephala														
Arabella iricolor		1	1						2	1				1
Arcturinoides sexpes														
Armandia spp												1	1	1
Ascidiacea spp														
Branchiomma nigromaculatum														
Bullia similis														
Byblis gaimardi		3		4	1	10		11	15	18	8	6	38	19
Capitella spp	183	42	1		9				1	1				
Capitellidae spp	1	6	2	1	2	5				3	1			
Caridea spp		2	2	3		10	4	1		3	2	10	1	
Chaetopteridae spp		2	1	1	17	1					2			1
Charybdis sp			1		1									
Chevalia aviculae						1	1							
Chlamys sp														
Cirolana spp					1		1	3	7	4	1			1
Cirratulidae spp	1			1	1				1					

Taxa	SWF3C	SWF4	SSAA	SSAB	SSAC	SSBA	SSBB	SSBC	SS1A	SS1B	SS1C	SS2A	SS2B	SS2C
Corophiidae spp	1	6	6	2	8	15	22	10	23	31	17	18	6	17
Cunicus profundus												1		3
Cyclaspis scissa														
Dehaanius spp					1									
Diopatra cuprea	4		1		2		1	1	2	1				
Diopatra neopolitana														
Diopatra spp		1	3		1		4	1						
Dispio magnus														
Donax spp		1												
Dorvillea spp										1				
Dromiidae spp													2	
Ebalia spp											1			
Echinocardium spp				1										
Echinoidea spp					2									1
Eocuma winri														
Epidiopatra spp														
Eunice spp		2				2	4	4	9	5	8	2	1	3
Eusiridae spp							1		15	6	5	14	6	3
Flabelligeridae spp									1				1	1
Galathea spp		2	5		7			8	11	6	3	7	3	4
Gammaridae spp														
Gammaropsis afra														
Gammaropsis spp	5	1	2		9	2	5	19	28	13	12			6
Glycera spp	10	15	7	8	8	10	1	1	7	7		2	2	
Glycinde capensis	1	5	2	8	5	5	1	2	3	3	7	3	7	3
Golfingia spp					1									
Gynodiastylis curvirostris														
Haminoea spp														
Harmothoe spp	1				1			1				1	1	
Hesionidae spp														
Hippomedon normalis		2	2									1	1	
Hippomedon onconotus			1	8		28	27	10	25	25	19	2		3
Inachidae spp														
Iphinoe crassipes			1			2	3	5	2		1	4	2	1
Lanocira gardineri									1					
Lasaeidae spp									1	1		1	1	
Leptochela spp					2	6		3	7	3	7	4	2	4
Leucothoidae														
Lumbrineris spp		1			1							1		1
Lysianassa spp														

Taxa	SWF3C	SWF4	SSAA	SSAB	SSAC	SSBA	SSBB	SSBC	SS1A	SS1B	SS1C	SS2A	SS2B	SS2C
Lysianassidae														1
Macoma spp							3		5	1	1			
Magelona cincta			1											
Maldanidae spp					1			1	2	2	1	1		1
Mandibulophoxus stimpsoni			2	10	2	19	24	21	17	16	31	4	1	9
Mysidacea spp						2	2		1			6		
Nassarius sp														
Natica gualteriana											1			
Nebalia capensis								1						
Nemertea spp			1			1							1	1
Nephasoma rutilofusca														
Nephtys dibranchus														
Nephtys hombergi	3	1	2	8		7	5	2	11	5	7	6	5	5
Nephtys sp					1									
Nereis sp	5	2			27			3	3	4	3	3	3	1
Nerinides spp	1													
Notomastus spp	1	1				1	1						1	1
Notophyllum splendens					1									1
Octocorallia spp														
Oedicerotidae spp		1										1		
Onuphis conchylega														
Onuphis eremita	2	2	1	7		3	6	5	8	7	3	1	2	2
Onuphis spp	1	1		1	2	2		1	2	3	6	1		1
Ophiuroidea spp					5							1		
Orbinia bioreti				1		2						2	1	
Orbinia cuvieri						2	2		3		3	1	2	3
Orbinia spp														
Orbiniidae spp						1			2					
Ostracoda spp	0	0	4	3	2	4	9	5	3	0	2	0	1	1
Owenia fusiformis									1					
Paguridae spp														
Paraonidae spp								1			1	1		1
Parvicardium turtoni								1				2		
Pectinaria spp											1			
Petricola sp						2								
Philine aperta												1		
Philyra globulosa		1											1	
Philyra spp														
Pholoe sp														

Taxa	SWF3C	SWF4	SSAA	SSAB	SSAC	SSBA	SSBB	SSBC	SS1A	SS1B	SS1C	SS2A	SS2B	SS2C
Photis spp			3	1	1									
Phyllodoce madeirensis			1				1		1					
Phyllodoce spp		1	1	1				1			1	1	1	1
Phylo spp	2	1	1		1								1	
Pinnotheres sp														
Platyischnopus herdmani						3	1	4	1					
Platylambrus quemvis														
Podocerus spp									2					
Poecilochaetus serpens	2					1	1	1	1	1				
Polydora spp			1							1	1		1	
Porcellana spp	1		1		12	3	3	1	1			1		
Portunidae spp	1										1	1		
Prionospio ehlersi										1				
Prionospio pinnata				1	2			1						
Prionospio saldanha														
Prionospio spp	6	2	8	6	4	5	10	7	6		5	9	8	8
Pseudojanira spp														
Pycnogonida sp														
Sabellidae spp		2	5	3	3	10	6	11	1	1	3		4	4
Schizammina pinnata							1					1	1	1
Scoloplos spp		1	1		1		1	1	3	1		1	1	
Semele spp														
Serpulidae spp														1
Sigalion capense									1	2	2	1		1
Sipuncula spp							1							
Spio spp		1				1	1							
Spiochaetopterus vitrarius		2	2	2	1	2		2		1	2	4		
Spionidae sp										2			2	
Spiophanes bombyx	1	1	2		3	1	1	2	5	3	2	2	2	
Spiophanes soederstromi		3		1			2		1	2	3		1	1
Sthenelais spp														
Syllidae spp			2		2									
Talitridae spp			1	1										
Tanaidacea spp											1	2	2	
Tellina spp														
Tellina vidalensis			1					6						
Terebellidae spp		1	1		2	1		1	17	6	1	1	1	
Tharyx spp														
Timoclea arakana				1	1	4	8	1	4	3	6	3	3	
Unciolella spp		6	19	8	5	151	172	146	156	296	204	76	29	125

Taxa	SWF3C	SWF4	SSAA	SSAB	SSAC	SSBA	SSBB	SSBC	SS1A	SS1B	SS1C	SS2A	SS2B	SS2C
Urothoe elegans			1		1	7	5	5	2	4				
Urothoe pinnata													1	
Urothoe pulchella		3	7	9		20	25	16	13	12	26	25	5	7
Xanthidae spp													2	

Chapter 6
Synthesis

6.1. Introduction

As discussed in Chapter 1 of this report, the Durban outfalls monitoring programme uses a weight of evidence approach to identify the impact of effluent discharge through the Central Works and Southern Works outfalls on the receiving marine environment. For this purpose, various physical, chemical and biological indicators of environmental condition are measured in water and sediment samples, and the toxicity of effluent discharged through both outfalls is tested. Although weighting criteria for the different indicators have not been defined, professional judgement is used to reach conclusions on the extent and magnitude of impacts.

Since the chapters of this report analyse and summarise the findings of different components of the monitoring programme separately, this chapter provides a synthesis of the main findings.

6.2. Synthesis of Main Findings

6.2.1. Effluent Toxicity

The toxicity of final effluent discharged through the Central Works and Southern Works outfalls was tested for two main reasons. First, it provides an understanding of temporal variability in the toxicity of effluent discharged through the outfalls. Second, the toxicity data can be used to estimate whether toxic effects are likely to manifest beyond the zone of dilution under worst case conditions (i.e. no current flow in the receiving marine environment).

On none of the 12 testing dates did the number of dilutions required to render final effluent from the Southern Works wastewater treatment facility non-toxic to sea urchin gametes exceed the lowest theoretical minimum initial dilution of the outfall that serves this facility. This indicates that there was little risk of toxicity beyond the zone of initial dilution for the outfall. The situation was different for the final effluent from the Central Works wastewater treatment facility. Final effluent could not be collected on four testing dates, because of problems experienced at the facility. On three of the eight dates that final effluent was tested, the number of dilutions required to render the effluent non-toxic to sea urchin gametes exceeded the lowest theoretical minimum initial dilution of the outfall that serves this facility. Thus, provided there was no current flow at the time, similarly sensitive organisms beyond the zone of initial dilution in the marine receiving environment may have experienced toxic effects. The probability that toxic effects manifested is probably low considering that it is highly improbable that the receiving marine environment is ever stagnant and the number of dilutions required to render final effluent from the Central Works wastewater treatment facility non-toxic to sea urchin gametes was not much higher than the lowest theoretical minimum initial dilution of the outfall.

The toxicity of final effluent from the Central Works wastewater treatment facility was often higher than the toxicity of final effluent from the Southern Works wastewater treatment facility. This was contrary to expectation considering that the Southern Works wastewater treatment facility receives a high volume of industrial effluent, which was expected to reveal a higher toxicity. Problems experienced at the Central Works wastewater treatment facility may be a reason for the higher toxicities recorded for final effluent from this facility.

6.2.2. Water Quality

Water quality monitoring in the proximity of outfalls serves two main purposes. First, measurements can be used to track the dispersion and dilution of effluent and thereby assess whether the outfall is meeting system design specifications. Second, measurements can be used to assess potential risks posed to organisms in receiving waters inside and outside of the zone of initial dilution, by comparing measured variables to water quality guidelines for the protection of aquatic ecosystem health. Regulatory authorities in many countries typically require that measured physico-chemical variables meet water quality targets at the boundary of the zone of initial dilution. The principal focus of the water column physico-chemistry component of the monitoring programme is to determine whether

water quality at the margin of the zone of initial dilution at the time of monitoring complies with South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment).

Of the various physical, chemical and biological variables measured *in-situ* at the margin of the zone of initial dilution for the Central Works and Southern Works outfalls, none showed anomalies that could confidently be attributed to effluent discharge. Faecal indicator bacteria counts provided the clearest effluent signal. None of the other indicators measured provided signals that could confidently be attributed to effluent discharge. None of the water samples was toxic to sea urchin gametes.

The values and concentrations of the majority of physical and chemical variables were compliant with the South African Water Quality Guidelines for Coastal Marine Waters (Natural Environment) (DWAF 1995). There were non-compliances for pH, dissolved oxygen concentrations and copper. For pH, the non-compliances were spurious because the upper limit for this guideline is too low and is routinely exceeded in nearshore marine waters off the KwaZulu-Natal coast. For dissolved oxygen, the non-compliance is linked to a natural oceanic phenomenon (stratification) rather than effluent discharge. The dissolved oxygen concentrations were also not so low as to be of ecological concern. Two copper concentrations were non-compliant with the relevant guideline. One of these concentrations was in fact very high. However, it is difficult to link these concentrations to effluent discharge, since they were detected in samples collected a substantial distance from the outfalls, and concentrations at most stations were below the method detection limit.

Trends for most physical and chemical variables were qualitatively similar to those for the 2009 and 2010 surveys of the Durban outfalls monitoring programme.

6.2.3. Sediment Quality

The most significant impacts of effluent discharge through deepwater marine outfalls are commonly evident in the benthic compartment of the receiving environment. The Durban outfalls monitoring programme places particular focus on this compartment of the receiving environment for several reasons. In contrast to the water column, where high temporal and spatial variability due to turbulence and mixing means that monitoring provides only a snapshot of conditions at the time of monitoring and contamination events may be missed, the benthic environment provides a far more conservative, spatially and temporally integrated measure of conditions. Conditions in sediment are more stable than the water column and because most contaminants are particle reactive they tend to accumulate in sediment rather than remain in solution. Contaminant concentrations in sediment are often orders of magnitude (up to a million times) higher than in the overlying water column.

The findings of the 2011 survey of the Durban outfalls monitoring provide clear evidence that the discharge of effluent has impaired sediment quality in the vicinity of the diffuser sections of the Central Works and Southern Works outfalls. The impacts were, however, more frequent and of a greater spatial extent and severity in the vicinity of the Southern Works outfall. Sediment near both outfalls was characterised by high faecal indicator bacteria colony forming unit counts. In fact, faecal indicator bacteria were detected at all sites, including the reference sites, providing evidence that effluent was impinging on the benthic environment across the study area.

Sediment near the Southern Works outfall and to a far lesser degree and extent at the Central Works outfall was enriched with particulate organic matter. This has presumably caused the higher chemical oxygen demand of sediment near both outfalls as compared to reference sites, although once again the effects were more pronounced near the Southern Works outfall. At the Southern Works outfall, the accumulation of organic matter and the associated chemical and probably also biological oxygen demand clearly exceeded the rate of re-ventilation of the sediment with dissolved oxygen. This is evident in the strong aroma of hydrogen sulphide and discolouration of the sediment. Hydrogen sulphide is produced by heterotrophic bacteria under anoxic conditions, while the discolouration of sediment is probably due to sulphide binding ionic metals. Metal-sulphide complexes usually impart a dark brown or black colour to sediment.

Grain size normalised concentrations of several metals in sediment near both outfalls were higher than for sediment from most of the reference sites and provides evidence that the sediment was metal contaminated. Of the wide suite of organic chemicals analysed, the use of many of which is regulated under the Stockholm Convention on Persistent Organic Pollutants, only polycyclic aromatic hydrocarbons were measured at concentrations exceeding the method detection limit, and then only at three sites in the immediate vicinities of the outfalls and at one reference site. Although there was evidence for the contamination of sediment in the immediate vicinities of the outfalls by metals and polycyclic aromatic hydrocarbons, comparison of metal and polycyclic aromatic hydrocarbon concentrations to sediment quality guidelines suggests there is theoretically little probability that these contaminants were adversely impacting benthic biota.

However, sediment porewater at the majority of outfall and reference sites was toxic to sea urchin gametes. The magnitude of toxicity at Central Works outfall and reference sites was low with the exception of a reference site situated 6000 m to the southwest of the outfall. The toxicity of sediment porewater at Southern Works outfall reference sites was comparable and relatively mild, but was high at sites situated in the immediate vicinity of the outfall. The cause of the toxicity could not be satisfactorily attributed to ammonia concentrations measured in porewater, nor to concentrations of metals and polycyclic aromatic hydrocarbons in sediment. There is a possibility that the high toxicity of porewater at some sites in the immediate vicinity of the Southern Works outfall may be due to the presence of hydrogen sulphide, which is highly toxic to most marine organisms.

The most significant impacts of effluent discharge on the chemical properties of sediment occurred within about 300 m of the diffuser sections of the outfalls, although some impacts did extend as far as 500 m from the outfalls.

There was little difference in trends for most of the indicators of environmental condition between the 2009, 2010 and 2011 surveys of the Durban outfalls monitoring programme. In other words, the impact of effluent discharge from the Central Works and Southern Works outfalls on the benthic environment in the study area has remained broadly comparable for the latter period.

6.2.4. Benthic Macrofauna

A critical end-point of any outfall monitoring programme is to determine whether effluent discharge is adversely impacting the ecology of the receiving environment. Little relevance can be attached to a particular degree of contamination of water and sediment unless it can be evaluated within a biological context. Whereas sediment chemistry can only provide a screening level assessment of potential effects, the monitoring of biological communities provides a direct measure of effects. The Durban outfalls monitoring programme places considerable emphasis on an analysis of the structure and composition of benthic macrofaunal communities as an indicator for measuring the ecological impact of effluent discharge. Benthic macrofauna are invertebrate organisms greater than 1 mm in size that live in or on the sediment.

The structure of marine benthic macrofaunal communities is influenced by many factors. These include abiotic factors, such as the sediment conditions, salinity and temperature, as well as biotic factors such as food availability, competition and predation. A major challenge in environmental monitoring is to distinguish between naturally occurring and anthropogenically induced changes to benthic macrofaunal communities. This is best achieved through comparison of communities from impacted sites to those from reference sites. While benthic community data has limitations, properly analysed they remain the most ecologically relevant line of evidence regarding possible impacts on the benthos.

Univariate and multivariate analysis of benthic macrofaunal community structure for the 2011 survey of the Durban outfalls monitoring programme provides clear evidence that the seabed near the Southern Works outfall is enriched with particulate organic material. Benthic macrofaunal community structure in close proximity to the outfall has been modified because of this enrichment. This is manifested by reduced biodiversity and an increased abundance of opportunist capitellid polychaetes.

Comparison with earlier surveys reveals a gradual increase of this effect over the past decade. While this impact is not considered to pose an immediate ecological threat, its expansion is cause for concern and should be accounted for in management considerations especially if effluent volumes were to increase.

The Central Works outfall appears to be operating within the assimilative capacity of the receiving environment. In 2011, however, some indication was given of impacts manifesting in benthic macrofauna in close proximity to the outfall. Benthic macrofaunal community response at affected stations appears to be similar to that at impacted stations near the Southern Works outfall and is likely the result of mild organic enrichment of the sediment. This is presently of little concern but requires close monitoring in future surveys.

6.2.5. How Far do Impacts Extend from the Outfalls?

Critical information that managers of wastewater treatment facilities require is an understanding of the spatial extent of effluent induced impacts in the receiving water. It is difficult to provide a definitive answer in this context for the Central Works and Southern Works outfalls since the spatial extent of impacts differs according to the indicator. If the presence of elevated faecal indicator bacteria colony forming unit counts in sediment is used as the indicator then the conclusion is that effluent discharge impacts extended across almost the entire study area. However, faecal indicator bacteria are in and of themselves not necessarily indicative of an adverse impact for the reason that they pose no risk to the vast majority of marine organisms. Faecal indicator bacteria are, however, used to provide an understanding of whether pathogenic bacteria and viruses are likely to be present in water and sediment, but these too appear to provide no immediate ecological threat.

Other indicators demonstrated that effluent associated impacts extend across the entire grids of stations spanning the diffuser sections of both outfalls, although the severity of the impact varied considerably across the grids of stations. In the case of the Central Works outfall, this conclusion is based largely on a comparison between concentrations of some metals and the structure and composition of the benthic macrofaunal community to the communities at reference sites. The majority of other benthic indicators provided little evidence for adverse impacts in the vicinity of the Central Works outfall. There was no evidence for impacts at the nearest reference sites. It must, therefore, be concluded that adverse impacts to the benthic environment in the vicinity of the Central Works outfall, notwithstanding the comments made above for faecal indicator bacteria, extend somewhere between 500 m to 2 000 m of the outfall. The sampling design does not allow for a finer resolution of the spatial extent of impacts, in other words, no samples were collected between the outermost stations of the grids and the nearest reference sites. Based on professional judgement, however, it seems unlikely that the impacts extend much more than 500 m from the outfall.

At the Southern Works outfall, adverse impacts also extend across the entire grid of stations spanning the diffuser section. The most severe impacts were evident within about 300 m of the outfall, but some impacts were still evident at stations situated furthest from the outfall (about 500 m). There was also some evidence for impacts at a reference site situated 1000 m to the north-northeast of the outfall. Again, this assessment is made notwithstanding the comments made above for faecal indicator bacteria.

In conclusion, it is impossible to identify the precise spatial extent of adverse impacts in the benthic environment in the vicinities of the outfalls using the existing sampling design. If it is considered necessary to define the spatial extent at a finer resolution, then the sampling design will need to be modified.

6.2.6. Why do Benthic Impacts Differ Between the Outfalls?

It will by now be apparent to the reader that the frequency, spatial extent and magnitude of effluent discharge induced impacts in the benthic environment in the vicinity of the Southern Works outfall is greater than at the Central Works outfall. This raises the question as to why such differences manifest. It is difficult to provide an answer in the absence of detailed supporting data, including current velocities. There is, however, indirect

evidence from the sediment grain size composition and sorting data that currents are weaker in the deeper waters that impinge on the Southern Works outfall diffuser section compared to shallower waters at the Central Works outfall. However, numerical modelling has shown that the number of initial dilutions achieved is usually not very different between the outfalls. Other contributing factors will include the larger volume and loads of contaminants and particulate organic material in effluent discharged through the Southern Works outfall.

6.2.7. Should the Durban Outfalls Monitoring Programme Consider Farfield Effects on a larger scale?

Some stakeholders have questioned whether the Durban outfalls monitoring programme should investigate the fate of effluent derived contaminants at a larger farfield scale, that is, beyond the bounds of the study area. They contend that currents may be (consistently) dispersing effluent derived material to depositional zones, where contaminants might be accumulating to levels that pose an ecological risk. Depositional zones are areas where the currents are so weak that suspended material (e.g. mud, particulate organic matter) settles from the water column. Since this material is of a fine-grained nature and usually has a charged surface, it provides a potentially large surface area for the adsorption and retention of particle reactive contaminants. Contaminants can also adsorb onto particulate matter in the water column and hence be transported to depositional zones.

The scientists that prepared this report are of the opinion that while there is a need to investigate the status of the marine environment beyond the Durban outfalls monitoring programme study area, such studies should not comprise part of the outfalls monitoring programme for several reasons. First, the nearest depositional zone is at the Thukela Banks, off the mouth of the Thukela River (see Flemming and Hay 1988), about 85 km and 98 km north-northwest of the Central Works and Southern Works outfalls respectively. Although there is probably no conceivable reason why effluent derived contaminants could not be dispersed as far as the Thukela Banks, the likelihood for significant transport over this distance seems low.

Second, metal concentrations measured in surface sediment samples collected from the Thukela Banks in the last 5 years (Carter 2006, CSIR unpublished data) has provided no evidence for metal contamination[8]. In fact, the concentrations of most metals in the samples that have been collected fall within baseline metal concentration models and baseline concentrations for KwaZulu-Natal coastal sediment. As far as the scientists that prepared this report could establish, there are no recent data pertaining to the concentrations of organic contaminants in sediment from the Thukela Banks, nor indeed from other areas on the coastal shelf beyond the Durban outfalls monitoring programme study area. It is therefore impossible to determine whether organic contaminants are accumulating in the farfield at these larger scales.

Third, even if a comprehensive study of the Thukela Banks and other areas of the KwaZulu-Natal continental shelf does reveal elevated concentrations of contaminants in sediment it will be nearly impossible to identify the sources of the contaminants. This could include effluent discharge, but could also include stormwater and riverine discharges transporting contaminants into the sea (e.g. pesticides from agricultural lands).

Lastly, there is no evidence that metals are accumulating in sediment from deep waters (about 50 m to 80 m) between the Durban outfalls monitoring programme study area and at least the Mdloti area. Metal concentrations measured in sediment collected from the dredged spoil disposal ground off Durban and at three reference sites at six monthly intervals between 2007 and 2011 by the CSIR has provided no evidence for metal contamination with the exception of very low level zinc enrichment of sediment from the spoil disposal ground in certain surveys. The concentrations of zinc were of a very similar order to that measured near the Southern Works outfall. Furthermore, between 2003 and 2008 the Durban outfalls monitoring programme sampling design included reference sites at Amanzimtoti and Mdloti, and bar a few anomalous concentrations of arsenic in one survey at Mdloti there was no evidence that sediment from these areas was contaminated with metals or

[8] Note that Carter (2006) reached a different conclusion, but a detailed analysis of the data suggests his interpretation was incorrect.

persistent organic chemicals.

Consequently, while there is a need to investigate the status of the marine environment beyond the Durban outfalls monitoring programme study area the objectives of studies that might be initiated and the organisations that should fund these studies requires careful consideration, especially considering that studies covering large areas of the shelf will be expensive. The scientists that compiled this report are of the opinion that studies of the shelf environment beyond the Durban outfalls monitoring programme study area should be collaboratively funded by state agencies, but with national agencies playing the leading role (e.g. Department of Environmental Affairs).

6.4. References

CARTER R (2006) Environmental investigation programme to determine potential cumulative effects of outfalls and discharges to sea off the KwaZulu Natal coast. Report prepared for Coastwatch Project, Wildlife and Environmental Society of South Africa (Durban).

DWAF (Department of Water Affairs and Forestry) (1995) *South African Water Quality Guidelines for Coastal Marine Waters. Volume 1: Natural Environment.* Pretoria, South Africa.

FLEMMING B and HAY R (1988) Sediment distribution and dynamics of the Natal continental shelf. In: *Coastal Ocean Studies off Natal, South Africa*, Schumann EH (Ed), Lecture Notes on Coastal and Estuarine Studies 26, Springer-Verlag, Berlin.

Glossary of Terms and Acronyms[9]

Abiotic factors	The physical, chemical and other non-living components of the environment that an organism lives in. These factors include all aspects of climate, geology, and atmosphere that affect ecological systems.
Acute toxicity	The discernible adverse effects induced in an organism within a short period of time of exposure to a chemical. For aquatic animals, this usually refers to continuous exposure to the chemical in water for a period of up to four days.
Acute toxicity test	A method used to determine the concentration of a substance that produces a toxic effect on a specified percentage of test organisms in a short period of time.
Adsorption/Adsorb	Bonding of chemicals onto the surfaces of suspended particles by way of physical, chemical and biological processes.
Aerobic	An environment where oxygen is present or a process that uses oxygen.
Aliquot	A sub-sample of the original sample.
Ammonia (NH_3)	A chemical combination of nitrogen and hydrogen that occurs extensively in nature. It is a water-soluble gas that behaves as a weak base. It can exert toxic effects on aquatic life.
Ammonium (NH_4^+)	The protonated form and conjugate acid of ammonia. It predominates under low-pH conditions.
Anaerobic	An environment where oxygen is absent or a process that does not use oxygen.
Analysis of Variance (ANOVA)	A statistical procedure used to compare the average condition between three or more treatments.
Anthropogenic	Made and/or introduced into the environment by humans, especially pertaining to contaminants/pollutants.
Aquatic ecosystem	All the living and nonliving material interacting within an aquatic system (e.g., pond, lake, river, ocean).
Assemblage	An association of interacting populations in a given habitat (e.g., an assemblage of benthic invertebrates on the ocean floor).
Assimilative capacity	The amount of contaminant load that can be discharged to a specific water body without exceeding water quality standards or criteria. Assimilative capacity is used to define the ability of a water body to naturally absorb and use a discharged substance without impairing water quality or harming aquatic life.
Bacteria	Bacteria are single-celled, small organisms that reproduce generally by fission. Some are pathogenic (cause disease), but most are free-living, with some being saprophytic (feed on dead or decaying organic matter).
Bathymetry	Bathymetry is the water depth relative to sea level. From bathymetry data, an understanding of the seafloor topography can be gained.
Benthic	Pertaining to the environment inhabited by organisms living on or in the ocean bottom.
Benthic invertebrate community	The assemblage of various species of sediment-dwelling organisms that are found within an aquatic ecosystem.
Bioaccumulation	The net accumulation of a substance by an organism as a result of uptake from all environmental sources.
Bioavailable	A substance in a chemical and physical form that allows it to affect organisms or be accumulated by them.
Bioavailability	Degree to which a chemical can be absorbed by and/or interact with an organism.

[9] This glossary of terms and acronyms was compiled from numerous sources, which are available from the CSIR on request.

Biomagnification	A measure of cumulative bioaccumulation from the media source through two or more steps in a food chain.
Biomonitoring	Biological surveys over time using consistent sampling and analysis methods for detection of changes in biological condition.
Biota	The living organisms within a habitat or region.
Biotic	Relating to life or living things.
Bivalve	Mollusc with a shell in two parts, hinged together (e.g. mussel).
Chlorophyll	Green pigments in plants, including algae, that play an important part in the chemical reactions of photosynthesis. A measurement of chlorophyll-*a*, one type of pigment, is commonly used as an indicator of the algae content of water.
Chronic Toxicity	The response of an organism to long-term exposure to a chemical substance. Among others, the responses that are often measured in chronic toxicity tests include lethality, decreased growth, and impaired reproduction.
Colony forming unit (cfu)	A unit (measurement) of density used to estimate bacteria concentrations in ocean water. The number of bacterial cells that grow to form entire colonies, which can then be quantified visually.
Coliform bacteria	A group of bacteria primarily found in human and animal intestines and wastes. These bacteria are widely used as indicator organisms to show the presence of such wastes in water and the possible presence of pathogenic (disease-producing) bacteria. *Escherichia coli* (*E. coli*) is one of the faecal coliform bacteria widely used for this purpose.
Community	Any group of organisms belonging to a number of different species that co-occur in the same habitat or area.
Concentration	The quantifiable amount of a substance in water, food or sediment.
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.
Control site	A geographic location that is far enough from a known pollution source (e.g., ocean outfall) to be considered representative of an undisturbed environment. Information collected within control sites is used as a reference and compared to impacted sites.
Crustacea	A group (Phylum) of marine invertebrates characterised by jointed legs and an exoskeleton (e.g. crabs, shrimps, and crayfish).
Cumulative effects	Effects on the environment resulting from actions that are individually minor but that add up to a greater total effect as they take place over a period of time.
CSIR	Council for Scientific and Industrial Research
Day Grab	A mechanical device designed to collect bottom sediment samples. The device consists of a pair of hinged jaws and a release mechanism that allows the opened jaws to close and entrap a 0.25 m^2 sediment sample once they touch bottom.
DEA	Department of Environmental Affairs
Demersal	Organisms living on or near the bottom of the ocean and capable of active swimming (e.g. flatfish).
Dendrogram	A tree-like diagram used to represent hierarchal relationships from a multivariate analysis where results from several monitoring parameters are compared among sites.
Detection limit	The lowest concentration level in a sample that can be determined to be statistically different from a blank sample.
Dissolved oxygen (DO)	The oxygen that is freely available in water. Certain amounts are necessary for life processes of aquatic animals. The oxygen is supplied by the photosynthesis of plants and by aeration. Oxygen is consumed by animals,

	plants, and bacteria that decompose dead organic matter and some chemicals.
Diversity	A measurement of community structure that describes the abundances of different species within a community, taking into account their relative rarity or commonness.
Department of Water Affairs (DWA)	Government agency mandated with the control of wastewater discharges to surface waters of South Africa. Previously the Department of Water Affairs and Forestry.
Echinodermata	A group (phylum) of marine invertebrates characterized by the presence of spines, a radially symmetrical body, and tube feet (e.g., sea stars, sea urchins, and sea cucumbers).
Ecosystem	An interrelating complex of plant and animal communities and their associated non-living environment.
Effective Concentration (EC)	A point estimate (statistically derived) of the toxicant concentration that would cause a quantal ("all or nothing") effect, such as death or lack of fertilisation, in a given time, for example, 96 hr EC_{50}.
Effluent	The discharge to a body of water from a defined or point source, generally consisting of a mixture of waste and water from industrial or municipal facilities.
Endpoint	A measured response of a receptor to a stressor. An endpoint can be measured in a toxicity test or in a field survey.
Enterococci	Any *Streptococcus* bacteria that inhabit the intestines of warm-blooded animals. In the intestines, e*nterococci* are normal and do not cause disease. They can be pathogenic if they enter tissues, the bloodstream, or the urinary tract.
Epifauna/Epibenthic animals	The animals that live on the surface of sediments or on and among rocks and other structures.
Eutrophication	A condition in an aquatic ecosystem where high nutrient concentrations stimulate blooms of algae (e.g. phytoplankton). Algal decomposition may lower dissolved oxygen concentrations. Although eutrophication is a natural process in the aging of lakes and some estuaries, it can be accelerated by both point and non-point sources of nutrients.
Far-field effects	Effects of an activity that are observed far away from that activity.
Grab sampler	A device that is used to collect surficial sediments through a scooping mechanism.
Guideline	A numerical concentration limit or narrative statement recommended to support and maintain a designated water use.
Gypsum	A white or colourless mineral (hydrated calcium sulphate) that is a waste product of the fertiliser production process.
Habitat	A place where the physical and biological elements of ecosystems provide an environment and elements of the food, cover and space resources needed for plant and animal survival.
Heavy metal	An imprecise term with no sound terminological or scientific basis, used loosely to refer to metals that are toxic.
Hyaline membrane	Distinct membrane which forms around a fertilised egg.
Hypoxia	The condition of low dissolved oxygen in aquatic systems (typically with a concentration < 2 $mg.l^{-1}$ but > 0.5 $mg.l^{-1}$).
Impact	A change in the chemical, physical or biological quality or condition of a waterbody caused by external sources.
Impact Site	A geographic location that has been altered by the effects of a pollution source, such as a wastewater outfall.
Impairment	A detrimental effect on the biological integrity of a water body caused by an impact.
Indicator	Characteristics for the environment, both abiotic and biotic, that can

	provide quantitative information on environmental conditions.
Infauna/infaunal animals	Those animals that live within the sediments of the sea floor.
Inorganic	Any compound lacking carbon.
Intraspecific variability	Differences between individuals of a single species.
Invertebrate	An animal without a backbone (e.g. a starfish, crab, or worm).
Lethal Concentration (LC)	Toxicant concentration producing death of test organism. For example, a 96 hr LC_{50} would be the test concentration killing 50% of exposed organisms after 96 hours of exposure.
Macrofauna	Epifaunal or infaunal benthic invertebrates that are visible with the naked eye. These animals inhabit soft-bottom marine habitats and are retained on a 1 mm mesh screen.
Minimum Acceptable Toxicant Dilution (MATD)	The dilution needed to render an effluent non toxic, or at least no different to the controls.
Megabenthic invertebrate (Megafauna)	A larger, usually epibenthic and motile, bottom-dwelling animal such as a sea urchin, crab, or snail. These animals are typically collected by otter trawls with a minimum mesh size of 1 cm.
Meiofauna	Small interstitial (i.e. occurring between sediment particles) animals that pass through a 1 mm mesh sieve but are retained by a 0.045 mm mesh.
Method detection limit	The minimum concentration of a substance that can be measured and reported with 99 percent confidence that the concentration is greater than zero.
Mollusca	A taxonomic group (phylum) of invertebrates characterized as having a muscular foot, visceral mass, and a shell. Examples include snails, clams, and octopuses.
Multivariate analysis	Statistical methods (e.g. ordination or discriminant analysis) for analysing physical and biological community data using multiple variables.
Near-field	Effects of an activity that are observed adjacent or close to that activity.
Nitrate	A compound containing nitrogen that can exist in the atmosphere or as a dissolved gas in water. Nitrates in water can cause adverse effects on humans and animals and act as a nutrient for plants.
Nitrite	An intermediate in the bacterial transformation of ammonia or ammonium to nitrate.
Nitrogen	A key nutrient for aquatic and terrestrial plants and occurring in various forms (NO_2^-, NO_3^-, NH_3, NH_4^+).
Normalise	Perform a data calculation in order to express results in terms of a reference parameter or characteristic.
Nutrients	Essential chemicals (e.g. nitrogen and phosphorus) needed by plants for growth. Excessive amounts of nutrients can lead to degradation of water quality by promoting excessive growth, accumulation, and subsequent decay of plants, especially algae (phytoplankton).
Ordination	A two-dimensional scatter plot, generated through multivariate community analysis, which depicts the relative taxonomic similarities amongst a group of faunal samples.
Outfall	Discrete location where quantities of water and/or waste are discharged into lakes, streams, or oceans, generally through a pipe.
Parameter	One of a set of properties whose values determine the characteristics of a waterbody. Examples include dissolved oxygen, temperature, and salinity.
Pathogen	An agent such as a virus, bacterium or fungus that can cause diseases in humans. Pathogens can be present in municipal, industrial, and non-point-source discharges.
Phi (ϕ)	The conventional unit of sediment size based on the log of sediment grain diameter. The larger the Phi number the smaller the grain size.
Phosphorus	An important nutrient utilized by aquatic and terrestrial plants.
Physicochemical	Measurement of both physical properties (e.g. temperature, salinity) and

	chemical determinants (e.g. metals and nutrients) to characterise the state of an environment.
Phytoplankton	Free-floating, single-celled, microscopic plants that live in water (also called unicellular algae). Can make the water appear cloudy or coloured.
Polychlorinated Biphenyls (PCBs)	A group of closely related and manufactured chemicals made up of carbon, hydrogen, and chlorine. Due to their non-flammability, chemical stability, high boiling point and electrical insulating properties they have wide industrial and commercial applications. PCBs can persist for a long time in the environment and they can bioaccumulate and biomagnify in aquatic food webs and are suspected of causing cancer in humans. They are an example of an organic contaminant.
Population	An aggregate of interbreeding individuals of a biological species within a specified location.
Pollution	The terms 'pollution' and 'contamination' are often confused. The term 'pollution' is clearly defined in several of the international conventions, but in everyday language the term is used in another sense. The Paris Convention, for instance, defines pollution as the introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as hazards to human health, harm to living resources and to marine ecosystems, damage to amenities or interference with other legitimate uses of the sea. On the other hand, 'contamination' is caused by substances not normally present in the marine environment (or present in higher concentration than normal) that do not apparently cause ill effects.
Polychaeta	A taxonomic group (Class) of, mainly marine, invertebrates characterised by having wormlike features, segments, and bristles or hairs. They are very variable in form and lifestyle and are good environmental indicators.
Primary treatment	The first stage of wastewater treatment involving removal of debris and solids by screening and settling.
Receiving water	A river, stream, lake or other body of surface water into which wastewater or treated effluent is discharged.
Reference toxicant	A chemical used to access the constancy of response of a given species of test organisms to that chemical. It is assumed that any change in sensitivity to the reference substance will indicate the existence of some similar change in degree of sensitivity to other chemicals/effluents whose toxicity is to be determined.
Replicate	Taking more than one sample or performing more than one analysis.
Salinity	A measurement of the amount of salt in water. Frequently reported as parts per thousand (i.e. grams of salt per 1 000 grams of water) and abbreviated as ppt, but technically has no measurement units.
Sediment	Mud, sand, silt, clay, shell debris, and other particles that settle on the bottom of rivers, lakes, estuaries, and oceans.
Shell hash	Sediment composed of shell fragments.
Sorting	The range of grain sizes that comprise marine sediments. Also refers to the process by which sediments of similar size are naturally segregated during transport and deposition according to the velocity and transporting medium. Well sorted sediments are of similar size (such as desert sand) while poorly sorted sediments have a wide range of grain sizes (as in a glacial till).
Species	A category of biological classification ranking immediately below the genus, comprising related organisms potentially capable of interbreeding. A species is identified by a two part name; the name of the genus followed by a Latin or Latinised un-capitalised noun agreeing grammatically with the genus name.

Species richness	The number of species per unit area. A metric used to evaluate the health of macrofauna and meiofauna communities.
Site	A sampling location within a study area or site, where physical, chemical, or biological sampling and/or testing occurs.
Trace metal	A metal found in low concentration, in mass fractions of ppm (µg) or less, in some specified source (e.g. sediment, tissue).
Trophic level	A portion of the food web at which groups of animals have similar feeding strategies.
Taxon (taxa)	Any group of organisms considered to be sufficiently distinct from other such groups to be treated as a separate unit (e.g. species, genera, families).
Thermocline	The zone in a thermally stratified body of water that separates warmer surface water from colder deep water. At a thermocline, temperature decreases rapidly over a short depth.
Total Suspended Solids (TSS)	Insoluble solids that either float on the surface of or are in suspension in water or wastewater. TSS is a measure of the amount of particulate matter in an aqueous sample. May also be referred to as suspended solids (SS).
Toxic	Poisonous, carcinogenic, or otherwise directly harmful to life.
Toxicant	A chemical capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death (e.g. pesticides and metals).
Toxicity	A measure of the impact on a chosen biological process or condition.
Turbidity	A measure of the clarity of water.
Upwelling	The process by which deep, cold, nutrient-laden water is brought to the surface, usually by wind divergence of equatorial currents or coastal winds that push water away from the coast.
Wastewater	Spent or used water of a community or industry, including runoff water and combined sewer overflow.
Water column	The area of water contained between the surface and the bottom of a waterbody.
Water quality guideline	A value, not to be exceeded, set for a specific water quality constituent in a defined water body portion or a water body, to ensure with a given measure of reliability, its agreed fitness for use. This is an achievable value determined by considering the water quality requirements of recognised water users as well as relevant physical, technological, economic and socio-political issues.
Whole effluent toxicity	The total toxic effect of an effluent measured directly with a toxicity test.
Zone of Initial Dilution (ZID)	An area in the immediate vicinity of a marine outfall discharge where there is rapid mixing of the effluent with sea water as a result of jetting and buoyant rise. An allocated impact area, or mixing zone, in a water body where numeric water quality criteria can be exceeded as long as acutely toxic conditions are prevented.